Mine and Mill Wastewater Treatment

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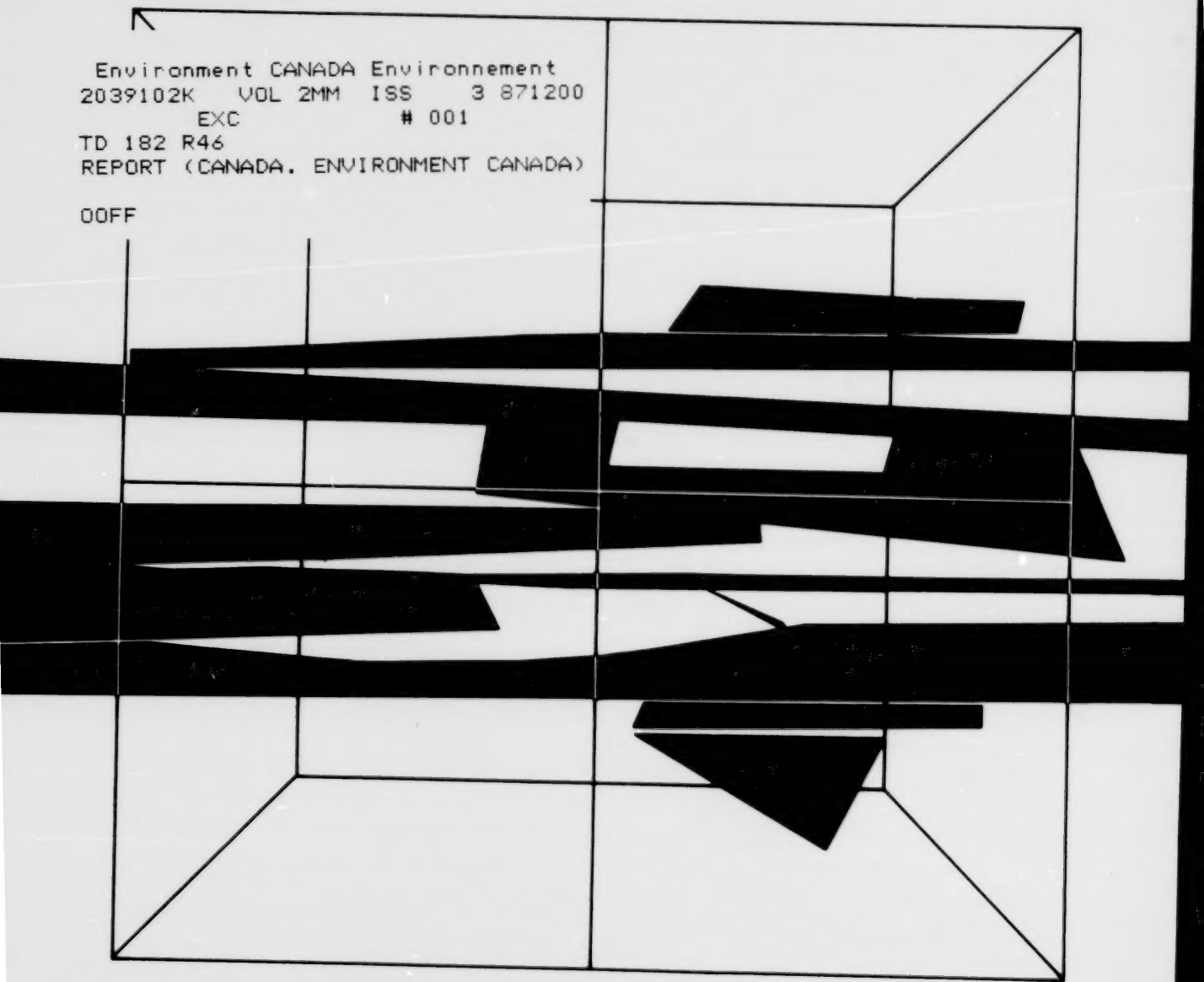

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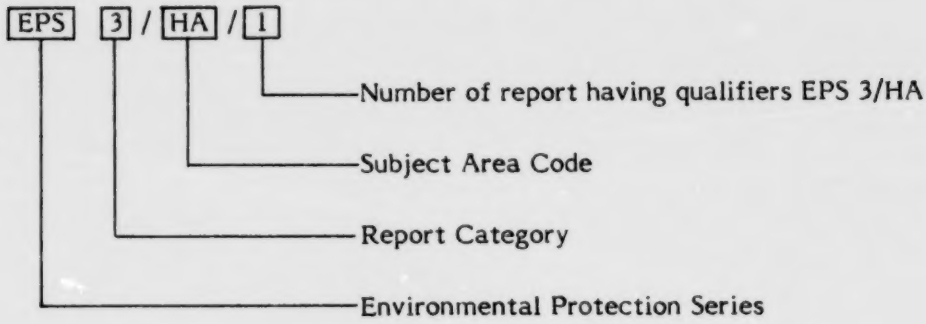

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MINE AND MILL WASTEWATER TREATMENT

by

Mining, Mineral and Metallurgical Processes Division
Industrial Programs Branch
Environmental Protection
Conservation and Protection
Environment Canada

Report EPS 2/MM/3
December, 1987

READERS' COMMENTS

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ABSTRACT

This report gives an overview of waterborne liquid wastes (mine drainage, mill process effluents and surface runoff) and their contaminants frequently encountered in metal mining and milling operations, particularly in the context of Canadian mining. Current wastewater management and contaminant removal practices, using both conventional time-proven tailings pond systems and newer mechanical-type treatment plants, are described.

The subject of acid mine drainage resulting from the mining of sulphide containing ores and the treatment of cyanide containing gold mill effluents are addressed extensively.

A final section in the report focuses on wastewater management during the post-operational phase of a mine's existence.

RÉSUMÉ

Ce rapport présente un survol des résidus liquides les plus importants (drainage de mine, effluents de procédés, ruissellement) ainsi que de leurs contaminants, rencontrés dans les opérations d'extraction et de préparation du minerai, particulièrement dans le contexte de l'industrie minière canadienne. On y décrit les méthodes actuelles de gestion et d'épuration des eaux résiduaires utilisant aussi bien le système conventionnel et bien éprouvé du parc à résidus, que les usines plus récentes de traitement de type mécanique.

Le drainage minier acide résultant de l'exploitation des minerais sulphurés ainsi que le traitement des effluents cyanurés des mines d'or sont décrits abondamment.

La dernière partie du rapport porte sur la gestion des eaux résiduaires durant la période post-opérationnelle de la vie d'une mine.

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1 INTRODUCTION

The Mine and Mill Wastewater Treatment Report (Report No. EPS-3-WP-75-5) was originally published in December 1975. At that time, it was the first available documentation to give guidance about mine wastewater management practices in Canada and has been a valuable reference to both industry and regulatory agencies for more than 10 years.

This report is an update of most of the topics presented in the original report, but also contains new material, particularly on gold milling and wastewater treatment at the post-operational phase of mining properties. It is intended to present an overview of wastewater treatment in metal mining and milling with specific focus on the origin of water pollution problems, wastewater treatment processes and actual wastewater management practices at mining and milling operations in Canada. It is not intended to present detailed coverage of the management of waste effluents discharged from metal mining and milling operations, nor is it meant to be a code of practice for the management of wastes for this industrial sector.

2 SUMMARY OF FEDERAL AND PROVINCIAL MINING EFFLUENT REGULATIONS AND GUIDELINES

2.1 Federal Requirements

The Environmental Protection Service of Environment Canada published "Metal Mining Liquid Effluent Regulations and Guidelines" (EPS 1-WP-77-1) in April 1977. The intent of the requirements defined in this document is to limit the discharge of deleterious substances from base-metal, uranium, and iron ore mines. Whereas mines existing at the time of publication are covered by guidelines; new, expanded, or reopened mines are subject to the regulations specified in the above document.

The requirements specified in Report EPS 1-WP-77-1 are uniformly applied national standards and intended to provide protection for fish and other aquatic life. (The regulations are promulgated under Sections 33 and 34 of the Fisheries Act). However, in specific locations, where they fail to protect the fishery resources, more stringent requirements may be imposed by the federal government through site-specific regulations. Where provincial requirements are more stringent than the federal regulations, the more stringent requirements prevail.

Authorized levels of substances specified as regulations or guidelines in Report EPS 1-WP-77-1 are listed in Table 1.

TABLE 1 AUTHORIZED LEVELS OF SUBSTANCES IN MINE EFFLUENTS

Parameter	Authorized Concentrations*	
	Max. Acceptable Arithmatic Monthly	Max. Acceptable In Grab Sample
Arsenic	0.5 mg/L	1.0 mg/L
Copper	0.3 mg/L	0.6 mg/L
Lead	0.2 mg/L	0.4 mg/L
Nickel	0.5 mg/L	1.0 mg/L
Zinc	0.5 mg/L	1.0 mg/L
Total Suspended Solids	25.0 mg/L	50.0 mg/L
Radium-226	0.37 Bq/L	1.11 Bq/L
pH	6.0 or greater	5.0 or greater

* All concentrations are total values except for Ra-226 which is at the dissolved level after filtration of the sample through a 3-micrometre filter. The levels shown apply to undiluted effluent at the point of discharge to the environment.

2.2 Provincial Requirements

Most provinces with metal mining operations within their jurisdiction regulate the industry case by case through the authority of various provincial statutes and regulations. Several provinces, however, use the limiting values defined in the federal regulations and guidelines as a basis for defining local requirements. The approach to regulating metal mines applied in each of the provinces is summarized in Table 2.

TABLE 2 SUMMARY OF PROVINCIAL METAL MINING REQUIREMENTS[a]

Province	Enabling Legislation	Summary of Requirements Relative to Federal Standards
Newfoundland	Dept. of Consumer Affairs and Environment Act	Environmental Control (Water and Sewage) Regulations specify general effluent requirements. These are generally consistent with monthly mean figures in the federal minimum requirements which are normally appended to mining permits.
Nova Scotia	Environmental Protection Act	Industrial Permits issued under Act specify adherence to federal requirements along with other site specific conditions.
New Brunswick	N.B. Clean Environment Act	Federal/Provincial accord ensures that federal regulations/guidelines are applied as minimum requirements.
Prince Edward Island	No metal mining	N/A
Quebec	Loi sur la qualité de l'environnement	Directive re: mining industry guidelines currently in draft form. Includes effluent requirements generally consistent with federal levels except copper which is 50% fed levels, Iron at 6.0 mg/L in grab sample, CN at 1.5 mg/L total and pH 6.5 - 9.5.
Ontario	Environmental Protection Act	Minimum provincial guidelines published for mineral industry. Ministry specifies a 'control point' at which these or more stringent project specific requirements are to apply.
Manitoba	Clean Environment Act	Federal requirements used as guideline in establishing site specific requirements.
Saskatchewan	Environmental Management and Protection Act	Federal requirements incorporated into specified limits for each project. Radium 226 specified as total rather than dissolved.
Alberta	No metal mining	N/A
British Columbia	Pollution Control Act	Pollution Control Objectives published for mining industry.

[a] as of March 1, 1986

3 THE ORIGIN OF WATER POLLUTION PROBLEMS

The origin of contaminated wastewaters at many Canadian mine sites can be largely attributed to the ease of oxidation of certain sulphide minerals with the consequent generation of acidic and metal-containing effluents.

3.1 Contaminants Due to the Nature of Ore Bodies

Although all metal sulphides may be amenable to at least slow oxidation, the formation of acidic effluents is associated with ore bodies containing the more readily oxidizable iron sulphides, such as pyrite (FeS_2) and pyrrhotite ($Fe_{0.8-1}S$). Thus, several mines in Canada, which contain the economic minerals (base metals, uranium, and precious metals) in a disseminated or massive iron sulphides matrix, generate acidic effluents.

3.1.1 Acid Mine Drainage and the Mechanism of its Formation. When fully established, this oxidative process results in the formation of acid mine drainage (AMD). As the name implies, AMD may be defined as a low pH leachate formed by natural processes acting on a sulphide-containing waste usually produced by a mining operation. Acidic effluents may arise as mine water from open pits and underground mine workings, or as surface drainage and seepage from tailings disposal facilities and waste rock dumps. Whereas the quality of AMD may vary from mine to mine, contaminated mine effluents often exceed regulatory standards for metals and acidity by several orders of magnitude. Hence, expensive treatment and control systems are required that can continue to operate for many years after commercial mining ceases.

However, many mines do not generate acidic effluents. The accurate prediction and successful prevention, treatment and control of acid mine drainage are key elements in minimizing the environmental impact of mining operations and require a thorough understanding of the mechanism of AMD generation.

The oxidation of sulphides present in ore bodies, tailings and mineralized waste rock generally occurs upon their exposure to oxygen (from air or from water). Pyrite, the most common sulphide mineral, is almost always associated with the generation of AMD. The oxidation of pyrite may be represented by four equations, although the complete process is complex and not fully understood.

$$FeS_{2(s)} + 7/2O_2 + H_2O \rightarrow Fe^{2+} + 2SO_4^{2-} + 2H^+ \quad (1)$$

$$Fe^{2+} + 1/4O_2 + H^+ \rightarrow Fe^{3+} + 1/2H_2O \quad (2)$$

$$Fe^{3+} + 3H_2O \rightarrow Fe(OH)_{3(s)} + 3H^+ \quad (3)$$

$$FeS_{2(s)} + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^+ \quad (4)$$

Kleinmann et al. (1981) described three stages in the generation of AMD.

Stage 1. Stage 1 involves the relatively slow chemical or biological oxidation of pyrite and other sulphide minerals near neutral pH (equation 1). This initial step may be catalyzed by the bacteria Thiobacillus ferrooxidans through direct contact with sulphide minerals. As acid begins to accumulate around the minerals, the process enters Stage 2.

Stage 2. Ferrous iron is oxidized to ferric iron (equation 2) which precipitates as ferric hydroxide (equation 3) and releases more acidity. As the pH falls even further, below about 3.5, ferric iron remains in solution and oxidizes the pyrite directly (equation 4).

Stage 3. In this last stage, the bacteria rapidly catalyze the process by oxidizing ferrous iron to ferric iron and the overall rate of acidity production is increased by several orders of magnitude (Singer and Stumm, 1970). Equations 2 and 4 combine to form a rapid cyclic process which produces large quantitites of acid associated with the release of heavy metals into solution. The resulting effluent is acid mine drainage.

Thiobacillus ferrooxidans can oxidize ferrous iron as well as inorganic reduced-sulphur compounds, whereas Thiobacillus thiooxidans oxidizes inorganic reduced-sulphur compounds only. These bacteria require an acidic environment for significant growth, although they may be capable of limited colonization at near neutral pH (Kleinmann and Crerar, 1979).

While the abundance of sulphide minerals is a key factor in the generation of acid mine drainage, other geochemical and physical factors also influence the process. As shown in the reactions, both oxygen and water are required reactants. Limiting the supply of oxygen limits the rate at which oxidation occurs (Pugh et al., 1984). Similarly, insufficient amounts of water may restrict oxidation rates in unsaturated systems (Lowson, 1982).

Water also plays a key role in removing the oxidation products from the surfaces of sulphide minerals. The frequency of flushings influences the quality of waters draining from mining waste. Acid and metal sulphate salts may accumulate within a waste deposit during relatively dry periods and be removed at times of higher precipitation (Geidel, 1980). AMD may be more concentrated with infrequent flushings. This is consistent with observed seasonal fluctuations in the quality of acid mine drainage.

Other factors, such as temperature, morphology of sulphides, particle surface area, availability of carbon dioxide, nutrients, and trace elements essential to the growth of microorganisms, also affect the rates of reactions -- and ultimately the quality of mine drainage.

Besides the abundance of sulphide minerals, however, the most important factor in determining the nature of mine effluents is often the presence of acid-neutralizing minerals. Carbonate minerals, such as calcite ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$), are the principal natural acid-neutralizing components of mining waste. These minerals can neutralize the acid produced by sulphide oxidation and therefore prevent the establishment of low-pH environments required by microorganisms. Under these conditions, the acid mine drainage process cannot become established and mine effluents remain uncontaminated.

The neutralization by calcite of acid produced by pyrite oxidation may be represented as follows (Williams et al. 1982):

$$FeS_{2(s)} + 2CaCO_{3(s)} + 15/4O_{2(g)} + 3/2H_2O \rightarrow Fe(OH)_{3(s)} + 2SO_4^{2-} + 2Ca^{2+} + 2CO_{2(g)} \quad (5)$$

Knowledge of the acid mine drainage process, and especially, the geochemical nature of mining waste are used to predict the formation of AMD.

Investigations into the mechanisms of acid generation have received considerable emphasis during the past 20 years and are of continuing worldwide interest. Selected references in this regard include:

Ainsworth and Blachar, 1984;
Buckley and Woods, 1984;
Groudev, 1983;
Hoffmann et al., 1981;
Kleinmann and Crerar, 1979;
Lowson, 1982;
Nordstrom, 1982;
Pugh et al., 1984;
Silverman, 1967;
Taylor and Wheeler, 1984;
Taylor et al., 1984;
Torma et al., 1974;
USBM, 1985;
Vuorminen et al., 1983;
Wakao et al., 1984, 1983, 1982.

3.1.2 Prediction of Acid Mine Drainage. The quality of mine water is determined by the rate of acid production and consumption and other rock-weathering processes. It is currently not possible to model all these processes to predict the quality of drainage accurately prior to mining. However, if the AMD-producing reactions become established, effluents of much poorer quality, compared to drainage from mines without

AMD, are produced. As such, the qualitalization by calcite of acid produced by pyrite oxidation may bative prediction of AMD is often the critical step in determining the impact a mine may have on the aquatic environment.

The potential for the generation of acid can be determined by laboratory testing of rock samples. Methods currently used for prediction have been recently reviewed (Perry, 1985; Ferguson, 1985; Ferguson and Erickson, 1986). Table 3 gives a summary of static and kinetic methods, including a short description, where they are used, their advantages and disadvantages, and key references.

Static tests. Several methods available to predict AMD determine the stoichiometric balance between acid-producing sulphide minerals and acid-consuming carbonates (Tests 1 and 2 in Table 3). These tests may be termed static, because they do not consider the relative rates of acid production and consumption. The acid-producing reactions are considered under kinetic control, whereas the acid-consuming reaction is an equilibrium process described earlier. Various investigators have criticized the use of static tests and suggested that they are only applicable to those cases where the quantity of acid-producing or -consuming material is in significant excess (Caruccio et al., 1980; Williams et al., 1982; SENES, 1984). However, Ferguson and Erickson in their review of AMD prediction techniques (1986), found that static tests were often accurate in predicting drainage quality and were particularly valuable as screening tests to determine whether more sophisticated procedures should be used.

Kinetic tests. Kinetic test methods attempt to model the acid-producing and -consuming processes in the laboratory. Kinetic tests, however, are not commonly used for AMD prediction because of their higher cost and longer duration, compared to static tests.

B.C. Research has developed a kinetic test which assesses the ability of bacteria to oxidize sulphide minerals in a rock sample (Test 3 in Table 3-1). This test is of particular value in determining whether a mineral constituent of the waste is toxic to the microorganisms and inhibits their growth.

For the remaining kinetic tests described in Table 3-1, columns have been used most frequently in Canada. In some cases, researchers have placed mixed rock types in a column to simulate tailings or waste rock structures. The effect of measures to prevent AMD formation can sometimes be tested in column or test plot experiments.

Mathematical models. Mathematical models have recently been developed to predict quantitatively mine drainage quality (Caruccio, 1984; Jaynes et al., 1984; Ricca and Shutz, 1979; Halbert et al., 1983; Nicholson, 1984; Cathles, 1979; Davis and Ritchie,

TABLE 3 ACID MINE DRAINAGE PREDICTION TECHNIQUES (adapted from Ferguson, 1985)

Test	Description	Use	Advantages	Disadvantages	References
Static Tests					
1. Acid/Base Accounting	1. Potential acidity is calculated from total or pyritic sulphur analysis. 2. Neutralization potential is obtained by adding a known amount of HCl and titrating with standardized NaOH to pH 7. 3. Potential acidity is subtracted from neutralization potential. A negative value below 5 tonnes $CaCO_3$/1000 tonnes of rock indicates a potential acid producer.	Extensively used in coal fields of Appalachia USA.	· Simple to use. · Low cost per sample. · Large number of samples can be tested to obtain estimate of variability in acid production potential. · Results in short time.	· Difficult to interpret if potential acidity and neutralization potential are in balance. · Does not consider possible different rates of acid producing and consuming reactions. · Several AMD controlling factors such as differences in particle size, temperature etc. not considered.	Sobek et al. (1978)
2. B.C. Research Initial Test	1. Acid producing potential is calculated from total sulphur analysis. 2. Acid consuming ability is obtained by titrating with standardized sulphuric acid to pH 3.5. 3. Acid producing potential is subtracted from acid consuming ability. A negative value indicates a potential acid producer.	Extensively used for base metal and gold mines in Canada.	as above	as above	Bruynesteyn and Hackl (1984)
Kinetic Tests					
3. B.C. Research Confirmation Test	1. Sample placed in 250-mL Erlenmeyer flask with 70 mL nutrient media and culture of T. ferrooxidans at pH 2.2-2.5. 2. Flask placed on gyratory shaker at 35°C in CO_2 enriched atmosphere. 3. pH monitored and additional sample added. 4. If pH rises then sample non-acid producer. 5. If pH remains low then sample potential acid producer.	Frequently used for base metal and gold mines in Canada.	· Relatively simple to use. · Relatively low cost. · Assessment of potential for biological leaching.	· May be difficult to interpret if pH does not strongly rise or fall. · Does not model initial AMD production step. · May take several weeks for sample pH to stabilize.	Bruynesteyn and Hackl (1984)

4. Shake Flasks	1. Sample placed in 1-L Erlenmeyer flask with 600 mL of water or nutrient solution. 2. Series of samples tested at various starting pH, innoculation, and temp. 3. Samples incubated for up to 3 months. 4. Leachates analyzed weekly and bi-weekly for range of parameters.	Infrequently used for base metal mines in Canada.	· Models acid production and consumption reactions kinetically. · Assesses effect of several controlling conditions for AMD production.	· May be difficult to interpret results. · Relatively high cost. · Large amount of data to compile and interpret. · Long time to complete test. · Not practical for large number of samples.	Halbert et al. (1983)
5. Soxhlet Reactor	1. Standard Soxhlet reactor used. 2. Water placed in reservoir, vaporized and passed into condenser. Condensed liquid dripped into thimble with sample and thence back into the reservoir. 3. Leachate analyzed after 64-192 hours.	Infrequently used in coal fields of Appalachia USA.	· Simple to use. · Results available in relatively short time. · Assessment of interaction between acid production and consumption reactions.	· Results may be difficult to interpret. · Relationship to natural processes not clear. · Test still in development stage.	Sullivan and Sobek (1982)
6. Humidity Cell	1. Sample placed in plexiglass container connected to humidified air. 2. Dry air passed over sample for 3 day, humidified air for 3 days and 200 mL of water added on seventh day. 3. Leachate removed and analyzed for range of parameters. 4. Procedure repeated for 8-10 weeks.	Relative common in coal fields of Appalachia USA.	· Models acid production and consumption reactions kinetically. · Attempts to model wet and dry cycle of environment.	· As for Shake Flask test.	Caruccio et al. (1980)
7. Columns	1. Sample placed in column and periodically leached by distilled water (usually for 8-10 weeks minimum). 2. Several variations of set-up in literature.	Relatively commonly used in Canada and US.	· Models acid production and consumption reactions kinetically. · Models effect of different rock types. · Can model wet and dry cycles of environment. · Considers different grain sizes up to model constraints.	· May be difficult to interpret results. · Relatively high costs. · Large amount of data to compile and interpret. · Long time to complete test. · Not practical for large number of samples. · May have difficulties with uneven leachate application and channelization.	Sturey et al. (1982) Ritcey and Silver (1981)
8. Test Plots/ Piles	1. Run of mine or modified sample placed on impervious surface. 2. Precipitation provides leachate which is collected in sump. 3. Samples of collected leachate analyzed for range of parameters. 4. Test usually run for at least 1 year.	Very infrequently used in Canada and US.	· Most realistic model of natural processes. · Can model relevent natural conditions in a single test.	· Very expensive. · Large amount of data to compile and interpret. · Very long time to completion of test. · Not practical for more than a very small number of discrete tests.	Eger et al. (1981) Murray and Okuhara (1980)

1983). These models are rather complex and have not been extensively verified or used for prediction to date.

3.1.3 Prevention of Acid Mine Drainage. Where testing procedures have identified that a mine could produce acid mine drainage, steps should be taken to prevent the process from establishing itself. However, the prevention of AMD has met with varying success. The difficulties are more associated with the nature of mining operations than a lack of understanding of the acid generation process. A mining operation with its extensive surface and subsurface ore bodies, mine workings, tailings and waste disposal areas poses obstacles to the control of sulphide oxidation.

Theoretically, the oxidation of sulphides can be prevented, or at least inhibited, by controlling the rate of transfer of essential reactants (air and water) to the sulphides or by blocking essential reactions of the acid generation process. While such methods can inhibit acid generation, complete elimination may often not be feasible (Harries and Ritchie, 1985; USBM, 1985; Monenco, 1984).

Minimizing the access of air to sulphides is regarded as the most effective measure for reducing the rate of acid generation. The complete elimination of air could be expected to stop the acid generation process. Minimizing the access of water (liquid phase) is much less significant in reducing the rate of acid generation, since the quantity of water to sustain the acid generation reactions is less than 1/5000 of the required quantity of air. It is significant, though, in limiting the transport of reaction products into the surrounding environment. However, the construction of diversion works, surface coverings, infiltration galleries, and waste compaction, all of which attempt to minimize the contact of water with potential acid-generating sulphide minerals, should reduce AMD contaminants released from a mine site.

To minimize the access of air, underground and open-pit mine workings are often flooded following completion of mining, especially where all the sulphides can be flooded and where convective movement within the water barrier is minimal. Water forms a very effective diffusion barrier to the transport of oxygen, thereby limiting the rate at which acid generation can occur (Ohio State, 1971; Smith and VanZyl, 1983). Flooding should occur as soon as possible after waste disposal to prevent oxidation. The use of flooding as a method of inhibiting acid generation within exhausted mine workings requires careful planning. In addition, many months of observation may be required, after flooding, to determine its effectiveness.

Effective inhibition of acid generation has been recorded for acid-generating tailings that have been deposited underwater, usually in lakes (Monenco, 1984; Hamilton and Fraser, 1978). The lack of acid generation can be attributed to the effectiveness of the water in limiting the rate of transport of oxygen to the tailings.

The difficulties in preventing acid generation within sulphide tailings deposits at Canadian mines have been recently reviewed (Monenco, 1984). Similar difficulties are associated with deposits of acid-generating waste rock and gossan. The access of oxygen and water to these deposits can be reduced by such methods as revegetation or covering with impervious materials. The alkalinity of groundwater can be increased substantially by a good mulch cover (Caruccio, 1984). The establishment of a vigorous vegetative cover may also reduce infiltration by increasing evapo-transportation. Their long-term effectiveness in preventing acid generation is debatable (Harries and Ritchie, 1985; Monenco, 1984; USBM, 1985). Often oxygen can diffuse through such cover materials as clays, gravels, and silts at a rate sufficient to sustain an appreciable level of sulphide oxidation.

Membrane liners such as PVC are very effective in preventing oxygen transfer, but their installation costs are high (Nicholas and Foree, 1982). Their long-term effectiveness has not been determined.

Various methods for blocking reactions responsible for acid generation are currently being researched. These methods include:

- application of bactericides to prevent or destroy the establishment of bacteria (Kleinmann et al., 1981; USBM, 1985, 1982); and
- addition of chemical reagents, such as phosphates, to remove ferric iron as an oxidant (Hoffmann et al., 1981; Nebgen et al., 1981; Watkin and Watkin, 1983).

Such methods, unless applied periodically, are likely to have application only as short-term inhibitors of acid generation until more permanent measures can be carried out.

In a few cases, sulphide minerals can be removed from the acid-generating environment. Sulphides can be separated from non-reactive minerals by conventional milling techniques or the entire waste can be placed in relatively inert environments such as underground mine workings. This technique is most applicable to small volumes of highly reactive wastes.

Where sufficient alkaline waste is available, potentially AMD-producing rock can be blended or segregated. These techniques have been used in the coal mining region of the United States with some success (West Virginia Surface Mine Drainage Task Force, 1979; Geidel and Caruccio, 1984). If blending is chosen, then it must be thorough to

ensure that no pockets of acid-generating materials are left in the waste (Smith and Sencendiver, 1981; Dollhopf, 1979). If segregation is chosen, then potentially acid-producing material should be isolated from the near-surface environment and material with excess alkalinity placed on its surface to promote revegetation (Infanger and Hood, 1980; Caruccio et al., 1980).

Implementation of an effective reclamation program as soon as mine waste structures have been completed is important in minimizing the potential for AMD formation.

The long-term success of preventing acid mine drainage is determined by the ability to establish a geochemically stable environment within the mine waste. In this environment, sulphide oxidation proceeds at such a slow rate that acid is either removed by infiltrating water or neutralized by alkaline minerals in the rock. Low pH micro-environments necessary for acidophilic bacteria do not exist and consequently, drainage water remains neutral and has a low metal content.

3.1.4 Treatment of Acid Mine Drainage. When prevention techniques have not been completely successful, acid mine drainage must be collected and treated. The location and volume of acidic leachates and effluents must be determined. Collection ditches or pipe systems must be constructed of sufficient capacity to handle occasional runoffs. Pumps, pipes, and fittings must be constructed of corrosion-resistant materials to prevent their destruction by the often highly acidic AMD.

The most common treatment of AMD is the addition of an alkaline reagent, such as lime, to neutralize the acid and precipitate heavy metals. A design manual has recently been published for chemical AMD treatment systems (EPA, 1983). These systems are effective, but capital and operating costs can be high. Chemical treatment systems often produce large volumes of sludge which must be properly handled (Ackman, 1982).

To reduce costs, passive treatment systems have been proposed which have lower capital and operating costs and require little maintenance. Permeable limestone barriers may remove some acid, but they are generally of limited effectiveness in removing copper and zinc because of their low operating pH (Wilmouth, 1974).

Soil may be effective in neutralizing acid mine drainage. Wangen and Jones (1984) found that the alkalinity of soil is the most important factor in removing metals from AMD. Crance et al. (1980) noted that the addition of crushed limestone substantially increased neutralizing capability of the soil. But soil can treat AMD at only

a limited rate, because of its low infiltration and neutralizing capability. Soil neutralization systems must be planned carefully, otherwise groundwater contamination can occur.

Recent studies have demonstrated the value of natural or artificial wetlands to remove heavy metals. Gersberg et al. (1984) noted a reduction in dissolved copper from 9.26 mg/L to 0.056 mg/L in spiked, secondary-treated municipal effluent after passing through an artificial cattail and bullrush wetland. The removal of 50 to 70 percent iron from an AMD stream using an artificial sphagnum moss wetland has been reported by Kleinmann (1985). Chan et al. (1982) state that cattails, reeds, and rushes may remove up to 0.35 kg of copper/hectare of wetlands. Holm and Jones (1985) describe an effective AMD treatment system that consists of an artificial peat bog and limestone trench. Total iron concentrations were reduced by 99 percent, and the pH was increased from 5.1 to 7.4.

The mechanism for the removal of metals by wetlands is not completely understood; it probably involves metal uptake by plants and metal sulphide precipitation by microbiological and chemical processes. Studies of passive treatment systems at low-cost, low-maintenance alternatives to conventional chemical acid mine drainage systems have increased in recent years (Burris, 1984; Weider and Sang, 1984). Serious consideration of these systems is necessary because AMD treatment may be required for many years after mining operations cease.

3.1.5 Control of Acid Mine Drainage. Though AMD can be readily treated, steps should be taken to reduce the volume and improve the quality of waters draining from mining wastes after the AMD process has established itself. Control of acid mine drainage would reduce both the annual cost and the duration of treatment. Many of the techniques applicable to the prevention of acid mine drainage are also relevant to its control; all attempts to control one or more of the basic elements of acid generation: oxygen, water, bacteria, and sulphide and carbonate minerals. Specific techniques include waste removal, surface coverings (impermeable layers), water diversions, infiltration galleries, revegetation, bactericides, carbonate surface covers, and water table elevation controls (Monenco, 1984; Russel et al., 1976; Caruccio and Geidel, 1985).

The advantages and disadvantages of selected techniques are listed in Table 4. In general, these approaches have met with varying success (see Table 5). Difficulties have been encountered in attempting to control the AMD process deep within a mine waste by only simple manipulations of the surface of the waste.

TABLE 4 SUMMARY OF ACID MINE DRAINAGE CONTROL TECHNIQUES

Technique	Description	Advantages	Disadvantages
1. Sulphide Removal a) enhanced biological recovery	Acid generation is promoted to reduce sulphide content of waste, leachate collected and metals recovered.	Almost complete removal of sulphides eventually. Cost recovery through metal sales.	The following requirements have to be met: · very careful control of leachates, because extreme contamination will occur; · treatment of final effluents; and · extensive maintenance and operation. The potential to increase problem by encouraging AMD reaction over wider area exists.
b) physical removal	Severe acid-generating zones are identified, excavated, and transferred to less environmentally sensitive location. It is effective if AMD remains localized.	Elimination of most critical problem areas.	It may result in increased short-term release due to disturbance of waste. A suitable disposal site must be found. The identification of severe acid-generating zones could be difficult.
2. Enhance Alkalinity a) limestone/lime applications	Surface covering of lime or limestone is placed on waste, providing alkalinity to infiltrating water.	Implementation is relatively simple, low in cost.	A significant long-term reduction of AMD is unlikely. It is difficult to calculate lime requirements and to ensure delivery to critical acid-generating sites.
b) alkaline trenches	Alkaline materials are placed in excavated trenches in principle recharge zones to mining waste.	This relatively low-cost technique is more effective than simple surface covering since alkalinity targetted to follow infiltrating groundwater. It requires little maintenance.	No guarantee of success. It may be difficult to locate and provide necessary alkalinity to most important groundwater seeps. Alkaline materials have to be replaced over the long term.
3. Water Control a) diversion of surface water	Water is diverted from upslope catchment to prevent inflow to mining waste.	Implementation is relatively simple, low cost. This technique should reduce volume and loading of AMD. It requires minimal maintenance.	AMD may not be completely eliminated because of water inflows via precipitation and groundwater.
b) recontouring and compaction of waste surface	With reshaping and compaction of the waste surface water is diverted from the surface. Special attention should be paid to the recharge zones.	Reduction of volume and loading of contaminants through limiting water and oxygen inflow. Little maintenance required.	Moderate cost; May be difficult to work on some wastes such as tailings; Unlikely to eliminate AMD completely; May initially increase AMD through generation of fine particles on surface of waste.

4. Oxygen Control a) capping waste with clay, plastic, grout, asphalt etc.	A cap is placed over the waste to prevent oxygen (and water) ingress.	Probably effective in significantly reducing AMD. Moderate maintenance required.	May be expensive for large waste dumps; May take time for reductions in AMD to be realized; Difficult to get air tight seal due to differential settlement, water erosion, and porosity of cap; Cover material may be difficult to find in large quantities; Moderate to high cost; May have initial flush of acidity and metals from previous oxidation; May be difficult to construct permanent diversions and dams; Difficult or impractical to flood high waste rock dumps.
b) flooding of waste	The groundwater or surface water is raised to establish a permanent water cover.	Probably effective in significantly reducing AMD in long term.	
5. Bacteria Control a) bactericides	Bactericides, such as sodium lauryl sulphate, sodium benzoate, or potassium sorbate, are spread on the surface of the dump to kill bacteria.	It is relatively simple to apply and can be effective in rapid reduction of AMD.	May not be effective, though little testing has been conducted to date at metal mines; Moderate cost; Difficult to ensure delivery to deep zones of waste; Concern for residual bactericides in leachate; Reapplications probably required.
b) phosphate rock	Phosphate rock applied to surface and depth of waste (phosphate precipitates iron breaking the ferrous-ferric-pyrite cycle).	May serve as self regulating control; Can bind sulphide minerals with coatings; May result in rapid reduction of AMD.	Relatively high cost; May not be effective, if not thoroughly mixed in waste; More applicable to prevention than control due to difficulty of application.
6. In-Situ Neutralization a) carbonate addition	Severe acid-generating zones identified, excavated, and acid-consuming carbonates thoroughly mixed in waste.	Self-regulating control; No further maintenance required.	May be expensive; May have only moderate success due to binding of limestone with coatings; May result in initial increase in AMD production due to disturbance of waste.

TABLE 5 EFFECTIVENESS OF SOME ACID MINE DRAINAGE CONTROL TECHNIQUES

Technique	Application	Result	Reference
1. Surface application of limestone.	Coal mine waste rock dump.	Ineffective.	Geidel and Caruccio (1982)
2. Regraded and surface application of limestone and revegetated.	Coal mine waste rock backfill.	Ineffective after one year (longer term benefits may become apparent).	Lusardi and Erickson (1985)
3. Regraded, construction of diversions, surface application of lime and sewage sludge, and revegated.	Gold mine waste rock or tailings.	Modest improvement after 10 years.	Hinkle (1984)
4. Regraded, surface application of bactericide, topsoil, lime and revegetated.	Coal mine refuse pile.	Significant reductions in acidity and metals after one year.	Shellhorn et al. (1985)
5. Construction of alkaline trenches (sodium carbonate and limestone).	Coal mine waste rock backfill.	Significant reduction in sulphate and acidity after one year.	Caruccio and Geidel (1984)
6. Flooded workings and construction of dry seals.	84 underground coal mines.	Flooded shafts generally better than dry seals. Seals generally reduced acidity and increased pH.	Bucek et al. (1976)
7. Surface application of lime, top soil and revegetated.	Metal mine waste rock dump.	Ineffective after 5 years.	Farmer and Richardson (1981)
8. Regraded, construction of clay seal, top dressed and revegetated.	Metal mine waste rock dump.	Significant reduction in oxygen and temperature after one year (effect on water quality unknown).	Harries and Ritchie (1985)
9. Control the infiltration of water to underground mine by regrading, compacting, construction of impervious seal and revegetation of ground surface.	Underground coal mines.	Effective (some control through AMD treatment).	Herrmann (1980)
10. Three test piles covered: a) topsoil, b) coarse sand, and c) coarse sand/sandy till. All piles revegetated.	Test piles of metal mine rock.	Minimal improvement in quality, but volume of leachate was reduced after two years for topsoil and coarse sand/sandy till piles. No reduction in coarse sand pile.	Eger and Lapakko (1981)
11. Construction of 11 block wet air seals and one double bulkhead seal.	12 underground coal mines.	Effective in reducing acidity, sulphate, and iron after 12 years.	Adams and Lipscomb (1980)
12. Regraded, installation of plastic liner, top soil applied and revegetated.	Coal mine waste.	Effective in reducing acid load after two years.	Caruccio and Geidel (1983)

In general, the control of AMD is more difficult than its prevention. Once established, the acid generation process is self-sustaining and difficult to stop. Internal heating from the exothermic reactions promotes air convection within waste dumps. Precipitation of metal sulphate salts, such as jarosite, serve as sources of metals and acidity long after sulphide oxidation has abated. In these cases, control of water inflow is of greater importance than oxygen reduction - often a difficult task. This highlights the importance of effective reclamation of mining waste as soon as equipment can gain access.

3.1.6 Research Needs. Much progress has been made in understanding the mechanism for the formation of acid mine drainage. Also the role of microorganisms in promoting acid generation is well understood.

Initial step in acid generation. What is not clearly understood is the initial step in acid generation, where the pH of micro-environments surrounding sulphide minerals is reduced from near neutral to levels favourable for bacterial growth. Knowledge of this initial step is critical in the prediction and prevention of acid mine drainage.

Prediction. Several methods of AMD prediction are in common use. However, very little verification work has been conducted. This work is necessary, if predictions are to attain good levels of confidence In the future, researchers may have a battery of verified AMD tests from which to choose an application appropriate for a particular mine.

Prevention. As AMD prediction becomes more accurate, effective prevention techniques will be incorporated into mine plans. In this way, mining could proceed at many sites with minimal acid generation. The type, effectiveness, and cost of the prevention techniques will depend on the potential for acid generation at a particular mine. As such, studies of prevention techniques are required for a range of potential AMD-producing mines.

Verification of AMD prevention techniques is also required. Studies of the environmental impact of underwater tailings disposal are necessary to demonstrate that this is an effective method of preventing acid generation and that advantages outweigh other potential environmental impacts, such as impairment of habitat for aquatic organisms.

Passive designs. Although conventional treatment systems are effective and well understood, passive designs hold much promise as the future of AMD treatment. Further research in this area is warranted.

Control. The control of AMD presents some of the greatest challenges to abatement. Techniques developed to date have had varying success, generally at high cost. Research is ongoing to demonstrate and improve the effectiveness of these techniques. New approaches may be developed in the future; at present, no completely effective, low cost, alternative exists. Effective control techniques are especially required for pits and waste rock dumps.

Increased effectiveness at reduced costs. Progress in understanding and abating the production of AMD has been remarkable in the last decade. Investigators now have access to a range of tools in the prediction, prevention, treatment and control of AMD. Much work remains, however, in improving the effectiveness and reducing the cost of all facets of AMD abatement, particularly for control techniques. The present pace of research indicates significant progress should be made in solving this difficult problem in the near future. Current research programs, by industry, government and universities, include the Reactive Acid Tailings Study (RATS) and National Uranium Tailings Program (NUTP) led by Energy, Mines and Resources Canada, and programs supported by the United States Bureau of Mines and West Virginia Acid Mine Drainage Task Force in the United States.

3.1.7 Radionuclides. In uranium ore mining and processing, radionuclides occur in both mine water and mill process wastewaters Mine water, for example, may contain significant amounts of radium, thorium, and uranium. In fact, the mine water at some mines contains sufficient uranium to make uranium recovery economical.

The processing of uranium ores by either acid leaching or carbonate leaching results in the occurrence of radioactive species in the process effluents. The natural decay of uranium (U^{238}), actinium (U^{235}), and thorium (Th^{232}) present in the ore produces radionuclides, notably radium-226, radium-228, lead-210, polonium-210, thorium-228 and thorium-230. In tailings pond overflows, radium-226 is the radionuclide of major concern because of its relatively high solubility and radiotoxicity, and further treatment for its removal is generally required.

The effectiveness of some control techniques for typical uranium mine and mill wastewaters are shown in Table 5.

3.2 Contaminants Due to the Processing of Ores

3.2.1 Process Reagents. Ore processing operations use various chemical reagents (see Table 6). These reagents or their residuals are directed to the various process

TABLE 6 TYPICAL ORE PROCESSING REAGENTS

Reagents	Comments
Acids (H_2SO_4, HCl, HNO_3)	--
Alkalis (CaO, $Ca(OH)_2$, $CaCO_3$, Na_2CO_3, NaOH, NH_4OH, NH_3)	--
Frothers and Collectors	Surface active organics.
Modifiers	Surface active organics and various inorganics such as NaCN, Na_2SO_3, $CuSO_4$, $ZnSO_4$, Na_2S, $AlCl_3$, $Pb(NO_3)_2$, silicates and chromates.
Sodium Cyanide	Used for precious-metal cyanidation and as a depressant for copper minerals in flotation processes.
Flocculants, Coagulants	Aluminum and iron salts, and organic polymers.

Source: Hawley, 1977

effluents as inorganic or organic contaminants. Inorganic contaminants consist of the reagents or their salts. The organic contaminants consist at least of traces of the various frothers, collectors, and modifiers used during flotation processes. In addition, mine waters may contain nitrogen-derived residuals resulting from blasting operations.

Cyanidation processes for the extraction of precious metals from ores result in wastewaters containing free and metal-complexed cyanide as well as other cyanide-bearing derivatives such as cyanate (CNO) and thiocyanates (CNS).

3.2.2 Suspended Solids. The processing of ores to recover economic minerals results in process effluents which contain suspended solids. The suspended solids may range from colloidal (non-settleable) to settleable materials. Similarly, mine waters and surface drainages in contact with ore, tailings, and waste rock usually contain suspended solids.

3.2.3 Trace and Other Elements. Effluents from the processing of ores can contain at least traces of many of those chemical elements associated with the particular ore body. Those elements most commonly noted as contaminants include aluminum, arsenic, cadmium, copper, iron, lead, manganese, nickel, zinc; and with uranium deposits, radium as well.

The extent to which any of these or other elements enter process effluents depends on the geochemistry of the ore body as well as the mineral recovery process. In general, flotation processes tend to result in minimal dissolution of minerals, whereas hydrometallurgical leaching processes can result in highly contaminated process effluents.

3.2.4 Thiosalts. Thiosalts are partially oxidized sulphur oxyanions which originate mainly in the grinding and flotation of a number of sulphide ores, particularly those containing pyrite and pyrrhotite, under alkaline conditions. The three thiosalt species of major concern include thiosulphate ($S_2O_3^{2-}$), trithionate ($S_3O_6^{2-}$) and tetrathionate ($S_4O_6^{2-}$). Thiosalts are of concern because they generate sulphuric acid as they become, with further oxidation, the more stable sulphate form.

The presence of thiosalts in tailing pond effluents, at concentrations of a few hundred milligrams per litre or less, can create serious environmental problems through their oxidation to acid in receiving waters, with accompanying lowering of the pH of the water. The acidification of some rivers and lakes in Canada has been attributed to the discharge of thiosalt-containing effluents from sulphide ore milling operations. The water pollution problem arises from the acid produced and not from the presence of the thiosalt species themselves. Also in some mills that treat complex sulphide ores by selective flotation and by extensive water recycling to the mill, thiosalts can result in metallurgical difficulties.

The mechanisms involved in the generation of thiosalts are not well-known. The initial sulphide mineral/water reaction to form thiosulphate appears to be one of the most complex, yet least understood, reactions. Based on more recent investigations (Rolia and Tan, 1985) in an alkaline medium, the sulphide minerals oxidize first to produce elemental sulphur, then to thiosulphate, polythionates, and finally to the sulphate form.

The extent of thiosalt generation during milling depends on many variables including mineral type and sulphide content, pH, temperature, grinding media, fineness of grind, aeration, agitation, dilution, alkalinity and flotation reagents, and residence time.

The formation of thiosalts in alkaline solution may occur according to the following reactions:

$$FeS_2 + 1/2O_2 + H_2O \rightarrow Fe^{2+} + 2OH^- + 2S^0$$

$$4S^0 + 6OH^- \rightarrow 2S^{2-} + S_2O_3^{2-} + 3H_2O$$

$$3\,S_2O_3^{2-} + H_2O + 2\,O_2 \rightarrow 2\,S_3O_6^{2-} + 2OH^-$$

$$4\,S_2O_3^{2-} + 2H_2O + O_2 \rightarrow 2\,S_4O_6^{2-} + 4OH^-$$

The oxidation of thiosalts to sulphuric acid occurs according to the following reactions (Wasserlauf and Dutrizac, 1982):

$$S_2O_3^{2-} + 2O_2 + H_2O \rightarrow 2H^+ + 2\,SO_4^{2-}$$

$$S_3O_6^{2-} + 2O_2 + 2H_2O \rightarrow 4H^+ + 3\,SO_4^{2-}$$

$$S_4O_6^{2-} + 7/2\,O_2 + 3H_2O \rightarrow 6H^+ + 4\,SO_4^{2-}$$

The rates of oxidation are highly catalyzed by bacteria (primarily by Thiobacillus ferrooxidans) and therefore influenced by temperature. These reactions are responsible for the pH depression problems when they occur in receiving waters of limited buffering capacity. Also, they offer the means to treat thiosalt-containing mill wastewater (see Section 4.4).

3.2.5 Wastewater Quality. Examples of wastewater qualities, for a number of mining sites in Canada are given in Table 7.

3.3 Wastewater Treatment Requirements

Wastewaters from the mining industry are treated to ensure that final effluents are acceptable for discharge into the environment. The acceptability is determined by provincial and federal regulations and guidelines, the assimilative capacity of the local environment, and by the availability of practical treatment technology. At present, wastewater treatment is practised in Canada mainly for pH adjustment and removal of:

suspended solids,
metals (copper, iron, lead, nickel. and zinc),
arsenic,
cyanides,
thiosalts, and
radium 226.

Treatment may also be required for the removal of other less common elements, such as antimony and mercury, which may be present at specific developments. Treatment processes for the removal of the key contaminants listed above are reviewed in Section 4 of this report.

TABLE 7 EXAMPLES OF WASTEWATER QUALITIES AT CANADIAN MINES AND MILLS

	Base Metal					Gold				Uranium	
Source of Wastewater	MW	MW	MW	MW+SD	TP	TP	TP	TP	TP	TP	TP
pH	2.6-3.0	2.4-2.8	2.5-4.0	2.8-3.3	3-5	7.6	8.2	7.7	7.8	6.7-10.6	4.8
Sulphate (mg/L)	1200			800-1800						490-1870	140
Cyanide (mg/L)						13.0	3.3	0.2	0.9		
Thiosalts (mg/L as $S_2O_3^{2-}$)					300-1000						
Aluminum (mg/L)			35-150								
Arsenic (mg/L)									0.1		0.13
Copper (mg/L)	5-12	10-20		1-52	<3.0	2.4	1.1	0.1	0.2		
Iron (mg/L)	500-1200	500-800	200-1000	24-230	10-200	0.6	8.2				
Lead (mg/L)	3-12	0.5-1.0	2-5	0.1-5.0	5-20	0.1	0.1		0.1		
Nickel (mg/L)						0.4	0.7	0.1			0.06
Zinc (mg/L)	1700	300-500	30-200	18-175	50-250	0.8	0.7	0.1	0.1		
Radium-226 (mBq/L)										7.4-38	6.7
Uranium-238 (mBq/L)											0.19

MW -mine water
TP - tailings pond overflow
SD - surface drainage

4 WASTEWATER TREATMENT PROCESSES

This section of the report describes the processes commonly used in Canada for the treatment of wastewaters from the mining industry. The application of these processes at a number of mines in Canada is described in Section 5.

4.1 Removal of Free Acid and Heavy Metals

4.1.1 Precipitation of Metal Hydroxides. The neutralization of free acid and the concurrent removal of metals by precipitation as hydroxides under alkaline conditions is the most common method of chemical treatment for wastewaters at both active and many non-active mines in Canada. The removal of metals as their hydroxides has proven to be effective and practical for the treatment of the often large volumes of wastewaters generated by the mining industry.

The hydroxides formed by many of the multivalent metals are characterized by decreasing solubility as the pH of the solution increases. This is shown in Figure 1 for a number of metals in these wastewaters. A number of metal hydroxides, however, become increasingly soluble as the pH, or hydrogen ion concentration, deviates in either direction from that pH at which the solubility is a minimum. This amphoteric nature of the hydroxides is due to the formation of soluble hydroxo-metal-ion complexes. Thus, the precipitated hydroxide would react with acids and bases as illustrated by the following equilibrium reactions for solid zinc hydroxide:

$$Zn(OH)_{2(s)} + H^+ \rightleftarrows Zn(OH)^+ + H_2O$$

$$Zn(OH)_{2(s)} + 2H^+ \rightleftarrows Zn^{2+} + 2H_2O$$

$$Zn(OH)_{2(s)} + OH^- \rightleftarrows Zn(OH)_3^-$$

$$Zn(OH)_{2(s)} + 2OH^- \rightleftarrows Zn(OH)_4^{2-}$$

The total solubility of zinc (Zn_T) at a specific pH is the sum of the free zinc ions and the soluble complexes, i.e.:

$$Zn_T = Zn^2 + Zn(OH)^+ + Zn(OH)_3^- + Zn(OH)_4^{2-}$$

It can be readily shown (Stumm and Morgan, 1981) that this relationship can be expressed as a function of the hydrogen ion concentration, i.e.:

$$Zn_T \rightleftharpoons K_1(H^+)^2 + K_2(H^+) + \frac{K_3}{(H^+)} + \frac{K_4}{(H^+)^2}$$

where each K is an equilibrium constant and (H^+) is the hydrogen ion concentration. The mathematical form of this relationship indicates that a minimum Zn_T exists at a particular (H^+) (or pH) and that Zn_T then increases for both lower (H^+) (higher pH) values and higher (H^+) (lower pH) values. Similar relationships can be derived for other amphoteric metal hydroxides.

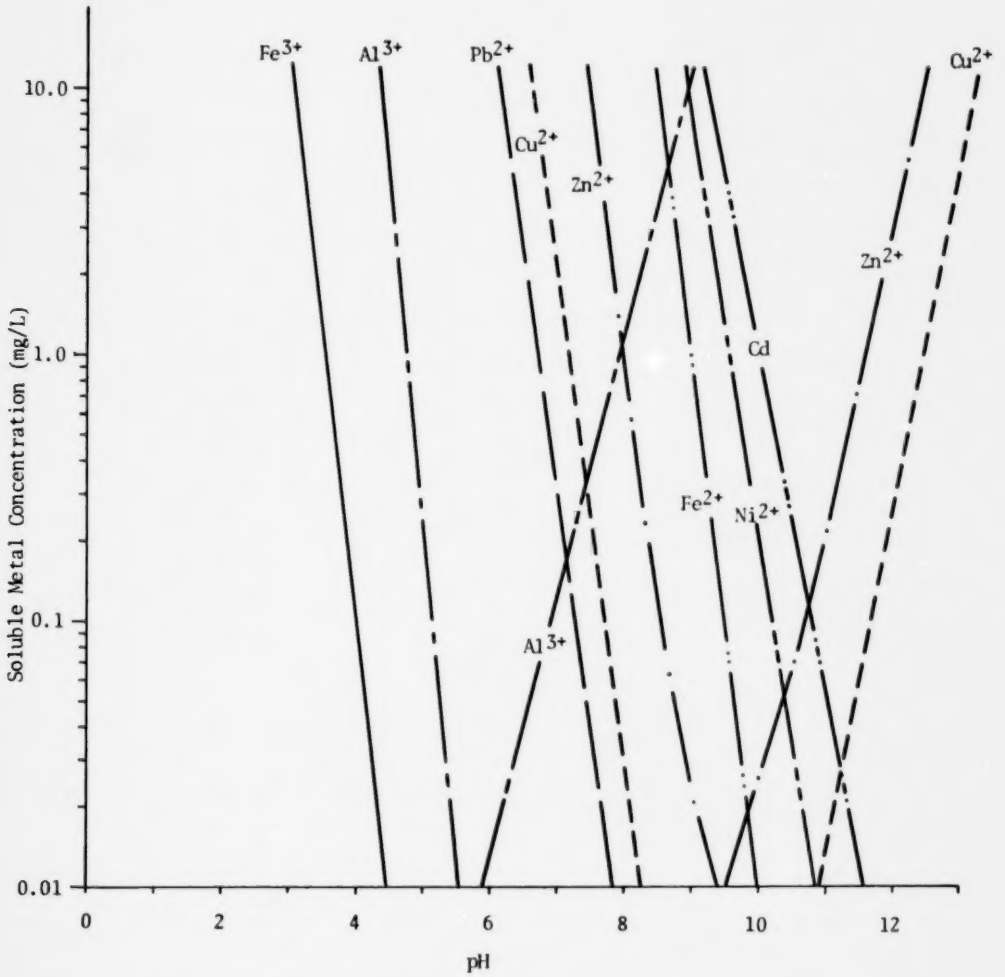


FIGURE 1 SOLUBILITY OF METAL HYDROXIDES

Other processes that occur during and subsequent to the precipitation of metal hydroxides contribute to the final concentrations of metals in solution. These processes, caused by the interaction between the precipitated solids and the dissolved species, include co-precipitation, ion exchange, surface adsorption, and occlusion (Jenke and Pagenkopf, 1983).

Multi-stage precipitation. Each of the metal hydroxides has a specific pH at which its solubility is minimal. This characteristic has been investigated as a means of obtaining a degree of separation among the various metals during their recovery from wastewaters as economic byproducts (Jenke and Diebold, 1983). The selection of the pH or pH's at which the metal hydroxides are to be precipitated depends on the composition of the particular wastewater. A wastewater containing aluminum and cadmium, for example, would likely justify the use of two-stage precipitation, the first at a pH of approximately 6 and the second at a pH of approximately 11 with solid-liquid separation between stages. Those metals most commonly present in wastewaters from sulphide mines in Canada include copper, iron, lead, nickel, and zinc. Effective precipitation of their metal hydroxides can be achieved at a pH ranging from approximately 9 to 10. Thus, single-stage precipitation of metal hydroxides is the common practice at Canadian mines.

Oxidation of ferrous iron. Ferrous iron, which normally occurs in acid mine waters, is oxidized during the precipitation of metal hydroxides to minimize the solubility of iron and ensure a more stable hydroxide precipitate (sludge). As shown in Figure 1, ferric hydroxide is considerably less soluble than ferrous hydroxide throughout the pH range. Hence, it is desirable to precipitate iron as the ferric hydroxide. The relative stability of ferric hydroxide at acidic conditions also ensures a more stable sludge.

As indicated in Table 8, ferrous iron is rapidly oxidized to ferric hydroxide by oxygen (in air) at the alkaline conditions used to precipitate metal hydroxides at Canadian mines. The oxidation of ferrous iron in alkaline solutions occurs according to the following overall reaction (Stumm and Lee, 1961):

$$Fe^{2} + 1/4\ O_2 + 2OH^- + 1/2\ H_2O \rightarrow Fe\ (OH)_{3(s)}$$

This reaction is sufficiently rapid so that the rate-limiting process is the rate at which oxygen can be transferred from the atmosphere to the wastewater. Thus, the capacity of the aeration system normally determines the rate at which ferrous iron is oxidized in alkaline solutions.

TABLE 8 OXIDATION RATES OF FERROUS IRON

pH	Reaction Half-Lives* (in minutes) at 10°C	at 20°C
7.0	216	22
7.5	7.0	0.7
8.0	2.2	0.2
8.5	0.07	0.007

* Time (in minutes) for one-half of ferrous iron to be oxidized by oxygen
Source: adapted from Stumm and Lee, 1961

The use of other chemical oxidants (ozone, hydrogen peroxide, chlorine, permanganates) or biochemical oxidation methods, which would allow for the more rapid oxidation of ferrous iron at acidic conditions, have not been considered in Canada.

Alkaline reagents. Hydrated lime ($Ca(OH)_2$), purchased as such or produced at the mine by the slaking of quicklime (CaO), is the most frequently used reagent for the precipitation of metal hydroxides at Canadian mines. Other reactive and strongly basic reagents, such as caustic soda (NaOH) and sodium carbonate (Na_2CO_3), can also be used to achieve the pH required for the precipitation of metal hydroxides. However, these reagents are rarely used for the treatment of mining industry wastewaters because of their relatively high cost. Relative costs for the various bases are given in Table 9.

In most situations limestone ($CaCO_3$) is not a suitable reagent because of its relatively low reactivity and alkalinity. These factors limit the pH that can be achieved to less than approximately 7. However, limestone combined with hydrated lime for final pH adjustment may be an appropriate reagent for the treatment of wastewaters which contain much free acid.

Scale formation and CO_2 absorption. The major anionic species in wastewaters from the mining and processing of sulphide ores is often sulphate. The use of a calcium-based reagent for the neutralization of free acid and the precipitation of metal hydroxides can result in the precipitation of gypsum, $CaSO_4 \cdot 2H_2O$ (K_{sp} 20×10^{-5}). The precipitation of gypsum increases the quantity of sludge generated by wastewater treatment. In addition, gypsum forms a concrete-like scale which may require periodic removal from the wastewater treatment equipment. The rate of formation of scale can

TABLE 9 RELATIVE COSTS OF NEUTRALIZING REAGENTS

Alkali	Cost of Reagent/Cost of Equivalent $Ca(OH)_2$
Calcium carbonate	0.3
Calcium oxide	0.8
Calcium hydroxide	1.0
Sodium carbonate	4.7
Sodium hydroxide	16

Source: Corpus Chemical Reports from 1985

be controlled to a certain extent by recycling of gypsum-bearing sludge to the neutralization system (high-density sludge process). The recycled sludge provides nuclei for the crystallization of gypsum from the supersaturated solution and improves settleability.

The absorption of atmospheric carbon dioxide into the alkaline system tends to lower the pH and encourage formation of those metal carbonates and metal hydroxide carbonates which are thermodynamically more stable than the corresponding metal hydroxides. This is a significant consideration for treatment systems using lagoons because of the increased surface area and residence time and consequent absorption of carbon dioxide.

4.1.2 Solid-Liquid Separation and High-Density Sludge Processes. The separation of precipitated metal hydroxides from the liquid phase is possibly the most difficult aspect of the treatment of mining industry wastewaters. Metal hydroxides precipitate as amorphous solids. This results in solids with low settling rates, poor sludge dewaterability, and a high degree of colloidal material which remains in suspension. Very effective separation of the precipitate from the liquid phase is essential to obtain minimum total concentrations of metals.

Various methods are used to improve solid-liquid separation and the dewatering of hydroxide precipitates. These include the recycle of sludge to the neutralization system (in mechanical treatment plants) to provide nuclei for precipitate formation and growth (high-density sludge processes) and the use of polymeric flocculants. The optimum degree of sludge recycle and flocculant type and dosage are determined by bench- and pilot-scale studies. Experience at Canadian mines, however, does indicate that adequate flocculation can be achieved by providing a thorough and rapid mix of the dilute polymer

solution with the neutralized wastewater just before it enters the solid-liquid separation system.

4.2 Removal of Cyanides

An overview paper which describes cyanide treatment methods applied to Canadian gold mill effluents was presented by Scott (1984). A more comprehensive review of the sources and treatment of cyanide-bearing wastewaters in the Canadian gold mining industry has been completed recently by Ingles and Scott (1987).

Wastewaters which contain cyanides are generated primarily as a result of the use of cyanide to extract gold and silver from ores according to the following reactions:

$$Au + 2\ NaCN + 1/4\ O_2 + 1/2\ H_2O \rightarrow NaAu\ (CN)_2 + NaOH$$
$$Ag + 2\ NaCN + 1/4\ O_2 + 1/2\ H_2O \rightarrow NaAg\ (CN)_2 + NaOH$$

The gold and silver are recovered from solution by either of two processes.

Merrill-Crowe process. In the original one, the Merrill-Crowe process, the precious-metal-bearing solution is filtered from the cyanided ore slurry and the gold and silver are precipitated from solution using metallic zinc dust, according to these reactions:

$$2NaAu\ (CN)_2 + Zn \rightarrow 2Au + Na_2Zn\ (CN)_4$$
$$2NaAg\ (CN)_2 + Zn \rightarrow 2Ag + Na_2Zn\ (CN)_4$$

Carbon-in-pulp process. In the carbon-in-pulp (CIP) process, the precious-metal complexes in the cyanided ore slurry are adsorbed directly onto coarse, activated carbon granules. The carbon granules are separated from the slurry by screening and treated in a separate, closed-loop operation to recover the gold and silver.

Both these processes result in wastewaters that are highly alkaline and rich in cyanide ions, stable metallocyanide complexes, and cyanide reaction products such as thiocyanate and cyanate. The extent to which any of these products occur depends mainly on the mineralogical composition of the ore, except for cyanate, which is an oxidation product of cyanide. Thus, in addition to free cyanide ions, the metallocyanide complexes of copper, iron, nickel, and zinc as well as thiocyanate, cyanate, and ammonia are commonly associated with the wastewaters. The cyanide content of the effluents is usually described, and characterized analytically, in terms of three classifications:

- free cyanide (CN_F),
- weak-acid dissociable cyanide (CN_{WAD}), and
- total cyanide (CN_T).

Free cyanide consists of ionic cyanide (CN^-) and hydrogen cyanide (HCN) only. Cyanide dissociable in weak acid includes those forms of cyanide less stable than iron cyanides, whereas total cyanide includes the iron cyanides as well.

The main differences between the waste effluents from the Merrill-Crowe process and the CIP process relate to the amount of cyanide contained in the mill effluents and the form of the effluent containing the cyanide. Most Merrill-Crowe mills use extensive barren solution recycle to take advantage of the residual leaching potential of the solution. However, a systematic bleed of the solution is necessary to avoid the buildup of deleterious compounds in the barren solution. As a result, the principal waste stream is usually smaller in volume than that from the CIP process, but carries proportionately higher concentrations of metallocyanide complexes and cyanide reaction products. In addition, it is in the form of a clear solution accompanied by a separate slurry stream consisting of repulped leached solids, which can contain a considerable amount of cyanide if the solids are not adequately washed with water. In the CIP process, the mill discharge is a slurry. There is neither filtration nor recycle.

Cyanide is also used at several base-metal mines as a depressant of copper minerals in the flotation of sulphide ores. The concentration of cyanide in these wastewaters is usually sufficiently low that treatment is not required. It is worth noting, however, that the first installation at a mine for cyanide destruction in Canada was at the Myra Falls operation of Western Mines Ltd. (now Westmin), a base-metal mine.

Gold mines in Canada use a greater array of methods for the removal of cyanide from their wastewaters than do those of any other country. The treatment methods currently in use in Canada are:

- natural degradation,
- alkaline chlorination,
- the Inco SO_2-air process,
- the Noranda SO_2 process, and
- hydrogen peroxide.

In addition, an ion-exchange cyanide removal and recovery process, that of CY-TECH Inc. of Mississauga Ont., is undergoing on-site pilot-scale testing at some Canadian mines. Still another process based on biological oxidation using microorganisms is in use at the Homestake Mine in Lead, South Dakota.

The following sections outline the principles underlying these treatment methods. Actual examples of plant application of these same processes, including flowsheets and pertinent recent operating data, can be found in Section 5.4.3.

4.2.1 Natural Degradation. The removal of cyanides by natural degradation requires prolonged exposure of mill effluents in tailings or other ponds to the atmosphere. It is the method most commonly used for cyanide removal at Canadian mines. The principal mechanism entails the loss of volatile hydrogen cyanide. At the pH of the effluents leaving the mill, most of the uncomplexed cyanide is in the form of non-volatile cyanide ions. On exposure to the atmosphere the alkaline wastewater absorbs carbon dioxide. As a result the pH drops toward 7. The dissociation constant for hydrogen cyanide (pKa) is 9.3 to 9.4, so that at pH 9.3, 50 percent of the cyanide is present as hydrocyanic acid (HCN), a value which increases to about 95 percent at pH 7.5. Under favourable conditions, this decrease in pH occurs within a matter of weeks and the volatilization of any uncomplexed cyanide proceeds at a corresponding rate. However, in many cases the bulk of the cyanide is in the form of metallocyanide complexes. Under these circumstances, the overall rate of volatilization of cyanide depends on the rate of dissociation of these complexes, which may require months. Exposure to ultraviolet light is a necessary factor in the case of the iron cyanide complexes; otherwise they do not dissociate at a significant rate. In addition, some ferrocyanide may be prepicitated by copper and zinc, as these metals are liberated by dissociation from their cyanide complexes. Regardless of the mechanism involved, significant removal of metal cyanides, including iron cyanides, occurs in the course of natural degradation.

Factors that also promote the volatilization of cyanide include warm ice-free conditions, large surface area-to-depth ratios, aeration, agitation, and exposure to light. While volatilization is by far the most important mechanism, other factors which contribute to the degradation of cyanide during prolonged exposure to the atmosphere include photo-decomposition, biodegradation, precipitation of sparingly soluble complexes, adsorption on solid surfaces, and slow oxidation to cyanate, bicarbonate and ammonia. Mathematical models that can be used to estimate the rates of natural degradation in tailings ponds are being developed. (Simovic et al., 1984). The principal inputs required are flow rates, concentration of cyanide in the influent, distribution of cyanide among the complexes, relevant environmental parameters, and pond dimensions.

The removal of cyanides by natural degradation is used at a number of mines in Canada (Ingles and Scott, 1987). Those mining sites where prolonged exposure of the wastewaters is economically feasible and results in acceptable effluent concentrations of cyanides will continue to use this method. Natural degradation is sometimes used for the partial removal of cyanide prior to other methods of treatment.

4.2.2 Alkaline Chlorination. Removal of cyanides from wastewaters by alkaline chlorination is common practice in the metal-plating industry. The method has been and continues to be used for the treatment of wastewaters generated by the gold mining industry in Canada to a limited extent (for example at Giant Yellowknife mine, described in Section 5.4.3). It is gradually being abandoned as cheaper and more effective methods are developed. The process relies on the oxidation of cyanide (CN) to cyanate (CNO), according to the following sequence of reactions at highly alkaline conditions (pH 10 to 11):

$$Cl_2 + H_2O \rightarrow \underset{\text{(hypochlorite)}}{OCl^-} + Cl^- + 2H^+$$

$$OCl^- + H_2O + 2e^- \rightarrow Cl^- + 2OH^-$$

$$2CN^- \rightarrow \underset{\text{(cyanogen)}}{(CN)_2} + 2e^-$$

$$(CN)_2 + Cl_2 \rightarrow \underset{\text{(cyanogen chloride)}}{2\,CNCl}$$

$$2CNCl + 4OH^- \rightarrow \underset{\text{(cyanate)}}{2CNO^-} + 2H_2O + 2Cl^-$$

The overall reaction for this process is $CN^- + OCl^- \rightarrow CNO^- + Cl^-$

The oxidation of most of the metallocyanide complexes occurs by a similar sequence of reactions. The metals precipitate as hydroxides as the complexes are broken up. One important exception is the ferrocyanide complex, which is instead oxidized only to the ferricyanide form and remains in solution. Other oxidizable species in the wastewater may be oxidized. In particular, thiocyanate is simultaneously oxidized to cyanate:

$$CNS^- + 4OCl^- + 2OH^- \rightarrow CNO^- + SO_4^{2-} + 4Cl^- + H_2O$$

The cyanate then slowly hydrolyzes to ammonium and bicarbonate as the pH of the treated effluent continues to fall. Further oxidation with chlorine at less alkaline conditions (pH 8.5) would lead to final conversion to nitrogen and bicarbonate:

$$3OCl^- + 2CNO + H_2O \rightarrow 2HCO_3^- + N_2 + 3Cl^-$$

The oxidation of the free cyanide and the complexes to cyanate is usually regarded as adequate treatment. Unfortunately, simultaneous oxidation of thiocyanate, if present, is unavoidable and can result in a substantial increase in reagent demand.

Alkaline chlorination is an effective method for the removal of free cyanide and those metallocyanide complexes less stable than the iron cyanides. However, it does not remove iron cyanides and that is a major disadvantage. In addition, the thiocyanate levels of many gold mill effluents are high enough to render alkaline chlorination very costly. Thus the effectiveness and economics of cyanide removal depend mainly on the composition of the particular effluent. Either chlorine gas or solutions of hypochlorites can be used as the reagents. In vessels equipped with agitators and instrumentation for the control of pH, the reactor residence times for alkaline chlorination are of the order of 30 to 90 minutes at pH 10.5 or higher. Any excess chlorine normally dissipates with time because of reactions with tailings solids and other oxidizable species. If retention time in the tailings pond does not permit this, provision must be made for removal of excess chlorine prior to discharge of the effluent to receiving waters.

4.2.3 Inco SO_2-Air Process. Inco Ltd. has patented a process for the removal of cyanides by oxidation to cyanate using sulphur dioxide (or a soluble sulphite) and air in the presence of copper ions as a catalyst (Devuyst et al., 1984). Copper(II) is used both as the catalyst and as the precipitating reagent for the elimination of ferrocyanide and more must be added, usually as copper sulphate, if an insufficient amount is present in the solution to be treated. The process has been and continues to be used at numerous Canadian mines. (Marchant, 1986; Zaidi et al., 1986). It has been the subject of intensive study, and there are increasing data on its capabilities and on the economics of its use. The oxidation of cyanide to cyanate occurs best at pH 8 to 10. The overall reaction is:

$$CN^- + SO_2 + O_2 + 2OH^- \rightarrow CNO^- + SO_4^{2-} + H_2O$$

The cyanide content of those metallocyanide complexes less stable than the iron cyanides is oxidized. The iron cyanides are not oxidized but precipitated as insoluble copper or zinc ferrocyanide compounds according to the reaction:

$$2Cu^{2+} + Fe(CN)_6^{-4} \rightarrow Cu_2Fe(CN)_6$$

The iron cyanide content of the effluents normally occurs as ferrocyanide, but even if it were to occur as ferricyanide, the oxidation-reduction potentials of the system are such that it is rapidly converted to ferrocyanide. Reagent consumption is reduced by avoiding the oxidation of thiocyanate, since this normally occurs at a much slower rate than that of cyanide. However, thiocyanate can be removed, if necessary, by using a different catalyst and modifying the operating conditions.

Reactor residence times, which are primarily determined by the rate at which air can be transferred to the wastewater, vary from approximately 20 to 90 minutes. The reactors are equipped with agitators and instrumentation for the close control of pH.

The SO_2-air process can be an effective method for the removal of total cyanides. The process is covered by a patent and its use is subject to a royalty. A flowsheet and operating data can be found in Section 5.4.3.

4.2.4 The Noranda SO_2 Process. Another sulphur dioxide process for the destruction of cyanide has been patented by Heath Steele Mines Ltd. (Ferguson; Walker, 1985) (The patent has been assigned to Noranda Incorporated.) This process, along with other processes, is undergoing evaluation tests at Noranda's Golden Giant Mine in the Hemlo area. It differs from the Inco process in that pure SO_2 is used. The injection of air as an oxidizer is not required for the removal of cyanide.

Pure SO_2 or undiluted industrial grade SO_2 is added at a rate that lowers the pH of the solution or slurry to between 7 and 9, followed by the addition of copper sulphate solution to produce an effluent whose cyanide content has been reduced to an acceptable level. No mechanism is described in the patent. A flowsheet and operating data are given in Section 5.4.3.

4.2.5 Hydrogen Peroxide. Interest in the use of hydrogen peroxide for cyanide removal has received considerable impetus as a result of its successful application by Degussa AG of West Germany at the Ok Tedi Gold mine in Papua, New Guinea (Knorre and Griffiths, 1984). As with the SO_2-air process, free cyanide and cyanide contained in those metallocyanide complexes that are less stable than the iron cyanides are oxidized, at a pH above 9.5 and in the presence of copper ions as catalyst, to cyanate. Ferrocyanide is neither destroyed nor oxidized to ferricyanide. Instead, as in the SO_2-air process, it is precipitated by copper. Thiocyanate is not oxidized and therefore does not consume any reagent. The process is now being used at two Canadian gold mines, and its application at a number of other mines is the object of current testing. Degussa AG has also developed sophisticated automatic control systems; their use is not essential though. The process is not covered by patents and no license fees are required for its use. Flow sheets and operating data can be found in Section 5.4.3.

4.2.6 Ion Exchange. The recent formation of a Canadian company to market technology and systems for cyanide recovery by ion exchange has revived interest in this method of treating gold mill effluents. Originally developed in South Africa (Goldblatt, 1956) and modified for use at Canadian gold mills by CANMET (Energy, Mines and

Resources Canada) (Gilmore, 1976), and by Cytech Inc. (Hussain, 1986), the process uses an anion exchange resin and is effective for the removal of all metallocyanide complexes. Uncomplexed cyanide ions are only weakly adsorbed by the resin and would not normally be retained, but this disadvantage is overcome by the addition of a small amount of a complexing reagent to ensure that only metallocyanide complexes are present. In the CANMET version, the metallocyanide complexes on the anion exchanger were broken up and eluted by an acid solution. The metals in this eluate were then recovered by a separate cation-exchange, absorption-elution cycle followed by neutralization and precipitation from the cation exchange eluate. The liberated cyanide in the anion exchange eluate was stripped by air and absorbed in a solution of sodium hydroxide for recycle to leaching. The metals were obtained as a metal hydroxide sludge.

The more recent Cytech version dispenses with the cation exchange column. Instead, the cyanide is stripped from the anion exchanger as hydrogen cyanide gas by a combined acid elution-air flushing step. The acidic eluting solution, which is passed upflow through the resin bed, desorbs and breaks up the metallocyanide complexes and at the same time regenerates the resin. The acidic eluate, containing the base metals and hydrogen cyanide gas in solution, is then passed downflow through a packed-bed scrubber column where it is contacted countercurrently with air. The hydrogen cyanide gas scrubbed from the acid solution is carried by air to a second packed-bed scrubber column. Here it passes upflow while contacting a countercurrent stream consisting of a solution of lime or caustic soda at pH 12. The caustic or lime scrubbing solution is recycled continuously to build up the cyanide concentration, while a periodic bleed returns the recovered cyanide to the leaching circuit. Similarly the acid eluate, after readjustment of its acid concentration, is also recycled continuously and a portion is bled off periodically for treatment in a neutralization-precipitation circuit. The precipitate of metal hydroxides may either be impounded in the tailings management area or possibly sold.

4.2.7 Biodegradation. Many microorganisms are capable of enzymatic oxidation or detoxification of cyanide. Treatment processes using microorganisms have been investigated sporadically since 1954. Only one such process, that at Homestake Mining Company's operation at Lead, South Dakota, is currently operating. It uses a mutant strain of Pseudomonas paucimobilis for the oxidation of both thiocyanate and cyanide to ammonia, followed by a second stage of nitrification-denitrification by indigenous, self-inoculated Nitromonas sp. and Nitrobacter sp. to complete the conversion to nitrogen.

Free and metal-complexed cyanides as well as ferrocyanides are all removed, the metals and ferrocyanide are absorbed by the bacterial biofilms, sloughed off, and filtered from solution.

The Homestake plant, which cost $10 000 000, uses rotating biological contactors (RBC) and treats 21 000 m^3 of combined mine and tailings impoundment water a day. It removes 85 percent of the total cyanide and 98.8 percent of the weak-acid-dissociable cyanide from a feed containing 0.5 to 10 mg/L of total cyanide and 45 to 75 mg/L of thiocyanate. However, temperature (10 to 18°C at Homestake) and feed concentrations are quite critical. Both the organism and the process are patented (Mudder and Whitlock, 1984).

4.3 Removal of Radioactive Contaminants

Wastewaters generated by the uranium mining industry in Canada are treated for the removal of radium by co-precipitation with barium sulphate. Barium sulphate is insoluble (K_{sp} $1x10^{-10}$) and readily incorporates radium into its crystal lattice because of the similarity in atomic structure of the two cations. Co-precipitation is achieved by the addition of soluble barium chloride to the wastewater to precipitate radium-barium sulphate which is removed in settling ponds or by filtration. The use of filtration to increase the effectiveness and reliability of solid-liquid separation is gaining acceptance.

For example, tailing pond decants are being treated at the Amok, Eldor and Key Lake uranium mines in northern Saskatchewan to below 0.37 Bq/L of radium-226 by a combination of chemical precipitation, pressure filtration, and settling. At Rio Algom's Stanleigh mine in Ontario, precipitation and gravity filtration is practised; no settling ponds are used as a subsequent step. At all four mines the recovered barium-radium sulphate precipitates are returned to the tailings ponds. The specification of filters and operating procedures must be carefully considered to ensure that scale formation within the filter bed (usually dual media) does not limit filter effectiveness and reliability.

4.4 Removal of Thiosalts

As already discussed in Section 3.2.4, thiosalts are of concern at certain sulphide ore mines because of their potential to generate acidic conditions within weakly buffered receiving streams. Acidification is a result of the slow oxidation of thiosalts to sulphuric acid. Thiosalts can be decomposed and removed from wastewaters as elemental sulphur, sulphides, or hydrogen sulphide gas. These and many other methods, such as chemical oxidation by hydrogen peroxide, chlorination, ozonation or sulphur dioxide-air,

electrochemical oxidation, reverse osmosis, reduction by iron, and adsorption by activated carbon have proven to be ineffective or impractical for application to the treatment of thiosalts in mining industry wastewaters (Wasserlauf and Dutrizac, 1983; Wasserlauf, 1985). At present, the potential for thiosalts to acidify receiving streams can be decreased or eliminated by biological oxidation in biodegradation ponds, followed by neutralization, prior to release of the effluent to receiving waters.

Thiosalts cannot be oxidized with air in the absence of a catalyst. So far investigations indicate that it is unlikely that a practical and economic system can be developed for the air oxidation of thiosalts (Wasserlauf, 1985). Both biochemical and chemical oxidation methods can convert thiosalts to sulphate at reasonable reaction rates. Biochemical methods rely on the thiobacilli, which oxidize reduced sulphur species under aerobic and acidic conditions, as a source of energy (Trudinger, 1967; Silver and Dinordo, 1981). Chemical oxidants such as hydrogen peroxide, chlorine, and ozone are all capable of the relatively rapid (15 to 60 minutes) oxidation of thiosalts to sulphate (Tan and Rolia, 1985). The costs for these reagents, however, limit their potential application to situations for which the operating costs would not be prohibitive. For example, the cost for these chemical reagents at a Canadian mine generating approximately 24.5 t/d of thiosalts (as $S_2O_3^{2-}$) would be approximately $5 to 14 million a year, depending on the particular reagent used (Wasserlauf, 1985).

The most promising process for the treatment of thiosalts continues to be biochemical oxidation. Extensive laboratory and pilot-plant studies have demonstrated that thiobacilli can remove thiosalts from mining industry wastewater (Silver, 1985; Wasserlauf et al., 1985). The rate of biochemical oxidation of thiosalts decreases markedly as the temperature falls below 20°C (see Table 10). The microorganisms show minimal activity at temperatures below 2°C. The rate at which thiosalts are oxidized also depends heavily on the population of microorganisms that can be established in the system. The significance of these factors has been demonstrated by the operation of pilot-scale aerated lagoons for 12 months (Wasserlauf et al., 1985). Thiosalts were not effectively oxidized using a conventional aerated lagoon system. The ineffectiveness can be attributed, at least in part, to the low population of microorganisms in the lagoon. A rock-filled aerated lagoon, however, was very effective for the removal of thiosalts from concentrations of 500 - 800 mg/L (as S_2O_3) to 100 - 200 mg/L. The hydraulic residence time provided by the 130-m^3 lagoon was approximately 80 hours; temperatures varied from approximately 1° to 20°C during the 12-month operating period.

TABLE 10 BIOCHEMICAL OXIDATION RATES OF THIOSALTS*

Wastewater Concentrations (mg/L) of Thiosalts (S_2O_3)	Residence time (days) to obtain Effluent Concentration of:					
	100 mg/L			200 mg/L		
	5°C	10°C	20°C	5°C	10°C	20°C
500	70	27	13	25	10	5
1000	150	60	30	67	27	13
2000	320	130	60	150	60	30

* Based on laboratory test results
Source: adapted from Silver, 1985

The practicality of constructing a full-scale treatment system using the rock-filled lagoon concept would require further consideration with respect to scale-up, potential for plugging of the rock-fill, start-up under cold weather conditions, and lag time between start-up and efficient operation. At present, various mining operations, including Brunswick Mining in New Brunswick and two Falconbridge operations in Ontario (Falconbridge and Moose Lake systems) rely on biodegradation of thiosalts by providing extended wastewater residence times (Norman and Michelutti, 1984). Thiosalt removal is effective during warm weather conditions but is less so during the winter and spring periods.

4.5 Removal of Arsenic

Wastewater from the processing of arsenic-bearing ores may contain varying amounts of arsenic in the form of arsenic(III) and (V) oxyanions, arsenite (AsO_2^-) and arsenate (AsO_4^{3-}). Arsenic can be very effectively removed from wastewaters by precipitation as either calcium or ferric arsenates. The presence of such metal ions as copper, lead, nickel, and zinc limit the solubility of arsenic because of the formation of sparingly soluble metal arsenates (Robbins, 1985). Arsenites can also be precipitated but they are generally more soluble and less stable than the corresponding arsenates. Hence, arsenite-containing wastewaters are often oxidized prior to precipitation to ensure that arsenate predominates. Air is an ineffective oxidant unless in the presence of a copper catalyst. Effective oxidants include ferric iron, hydrogen peroxide, chlorine, sulphur dioxide-air, and permanganate.

At highly alkaline conditions (pH 11 to 12), arsenic is precipitated as basic calcium arsenate ($Ca_3(AsO_4)_2 \cdot Ca(OH)_2$) by the addition of hydrated lime. This compound is very insoluble at highly alkaline conditions, so that dissolved arsenic concentration of less than .5 mg/L can be achieved. The use of this process, however, has been questioned (Robbins and Tozawa, 1982; Nishimura et al., 1985) because of the instability of the precipitate when disposed of in lagoons or buried in a landfill. The instability of the precipitate and dissolution of arsenic occurs as a result of the absorbtion of atmospheric carbon dioxide into the precipitate to form calcium arsenate:

$$Ca_3(AsO_4)_2 \cdot Ca(OH)_2 + CO_2 \rightarrow Ca_3(AsO_4)_2 + CaCO_3 + H_2O$$

Calcium arsenate ($Ca_3(AsO_4)_2$) is much more soluble than the basic calcium arsenate ($Ca_3(AsO_4)_2 \cdot Ca(OH)_2$), thus giving rise to the possible release of arsenic (Tozawa et al., 1985).

Arsenic can be precipitated from wastewaters by the addition of ferric iron to form extremely insoluble basic ferric arsenates ($FeAsO_4 \cdot xFe(OH)_3$).

The stability and solubility of these arsenates depends on the ratio of iron to arsenic, the larger the ratio, the more insoluble and stable the precipitate. Thus, whereas ferric arsenate ($FeAsO_4$) is relatively soluble, the basic arsenates with an iron-to-arsenic molar ratio of 8 or more are orders of magnitude less soluble in the pH range of approximately 2 to 8 (Krause and Ettel, 1985). Dissolved arsenic concentrations of 0.5 mg/L and less can be obtained by precipitation with ferric iron.

The precipitation of insoluble ferric arsenates is very likely accompanied by co-precipitation and adsorption processes involving interactions between the various arsenic species and ferric hydroxide precipitate (Krause and Ettel, 1985). Ferric and other multivalent metal hydroxides, such as aluminum hydroxide, are very effective for the removal of traces of contaminants. Thus, arsenic and many other elements, such as antimony and molybdenum (USBM, 1982; Ramadorai, 1978; Legendre and Runnells, 1975) can be reduced to levels of less than 0.5 mg/L by contact with ferric hydroxide. The process normally involves the addition of a soluble ferric salt to the wastewater followed by the addition of sufficient base to induce the formation of insoluble ferric hydroxide. In many situations, the wastewater contains adequate iron, thus only the addition of a base is required to induce the precipitation of ferric hydroxide.

4.6 Disposal of Treatment Sludges

The precipitates and other sludges from the treatment of mining industry effluents contain metals and other contaminants removed from the wastewaters. These

sludges are disposed of with mill tailings in separate disposal lagoons or by burial in on-site landfills, and in some situations on gravel drainage beds or in underground and open-pit mine workings. Whether the sludges are mechanically dewatered prior to disposal depends on the particular situation. All such sludge disposal sites are a potential source of pollution due to hydrological, hydrogeological, and geochemical processes. Hence, their design, operation, and eventual abandonment requires very careful consideration.

Most of the metal hydroxide sludges generated at Canadian mines are intermixed with tailings and disposed of in tailings ponds. Little information is available concerning the long-term chemical stability of the metal hydroxides in these situations. There is evidence, however, that gypsum-rich and alkaline hydroxide sludges, when disposed of in dedicated sludge lagoons and gravel drainage beds, tend to become more stable with aging due to the absorption of carbon dioxide. Absorption of carbon dioxide may result in the formation of the more acid-resistant metal carbonates as well as a concrete-like calcium carbonate-gypsum matrix, which is resistant to dissolution. Similar processes may occur in alkaline tailings ponds.

The stability of radioactive radium-barium sulphate sludges has also recently been investigated (Huck, 1985). The stability of these sludges, as would be expected, is controlled by the solubility of barium sulphate and the oxidation-reduction potential of the environment. The highly insoluble nature of barium sulphate favours its stability. However, the chemical or microbiological reduction of the sulphate species can lead to dissolution and release of radioactive radium. Hence, disposal sites which provide an aerobic environment are favoured. The current practice for the retention of these sludges at Canadian uranium mines is burial in tailings ponds.

4.7 Tailings Disposal

Canadian practices for the disposal of tailings are described in Section 5. As a general comment, tailings disposal ponds have proven to be very effective for the removal of suspended solids. In some situations, additional sedimentation capacity has to be provided or the solid settling characteristics have to be improved with coagulants.

4.8 Stabilization of Flotation Reagents

Many organic-based flotation reagents are potentially toxic to aquatic organisms (Hawley, 1977). These reagents, however, have rarely been known to cause toxic effects in receiving streams. The absence of toxic effects is due to their use in only limited quantities within mill circuits, adsorption and removal with concentrates, adsorp-

tion to the solids which remain within the tailings disposal area, and natural and biochemical degradation within the tailings disposal area.

4.9 Sewage Disposal

The sewage generated by a mining operation can be equivalent to that of many villages, hence sewage disposal is an important consideration. Systems most commonly used include waste stabilization lagoons, aerated lagoons, package treatment units, and direct discharge into the tailings pond where sufficient residence time can be provided to ensure adequate destruction of coliform and other enteric bacteria.

5 CANADIAN WASTEWATER MANAGEMENT PRACTICES

At most mines, the following three wastes streams have to be considered:

- mine water;
- mill process effluent including tailings; and
- contaminated surface drainage and seepages.

The objective of this part of the report is to review the manner in which the treatment processes described in the foregoing section are applied under practical conditions across Canada to achieve an effluent complying with provincial and federal regulations and guidelines (see Section 2).

All mine/mill operations have to dispose of large quantities of finely ground tailings usually representing 90 percent or more of the solids processed in the mill. Since the process of tailings disposal tends to be the predominant waste management function at many sites, it is appropriate to review first the alternatives available and the manner in which other treatment processes can often be integrated into the tailings disposal system.

5.1 Tailings Disposal

Factors influencing the choice of tailings disposal method include the following:

- chemical stability of the tailings;
- size classification of the tailings;
- local physical features including topography and availability of lakes;
- suitability of mining method and tailings characteristics for use of the coarse fractions as mine backfill;
- local environmental sensitivities with respect to flora, fauna, water quality, and aesthetics;
- availability of local material for construction of containment structures and final reclamation measures; and
- long-term, post-operational impact of the disposal area after reclamation measures have been applied.

At the design stage of a new mine, each of the tailings disposal methods listed in Table 11 should be evaluated and their relative advantages weighed with respect to the criteria listed above. Methods (1a) and (1b) in Table 11 are by far the most common approaches to tailings disposal used in Canada.

TABLE 11 TAILINGS DISPOSAL METHODS USED IN CANADA

Disposal Method	Comments
1. Discharge as a slurry into specifically designed containment area;	Most commonly used practice, Many configurations available depending on specific characteristics of site and tailings, Reclamation difficult in case of acid-generating tailings, Tailings pond can often be used for other treatment functions;
Subcategories include: a) Discharge into tailings pond (subaqueous discharge)	Produces lower in-situ densities than sub-aerial methods but has advantages in inhibiting transfer of oxygen to acid generating tailings;
b) Sub-aerial discharge methods onto tailings beaches including point discharge, spigotted, cycloned or layered tailings deposition	Allows coarse fraction to be used for dam construction, Can reduce permeability of deposited tailings, Can provide well drained accessible surface;
c) Disposal as thickened tailings in free-standing, mounded deposit;	Provides reduced disposal volume, Used at only two mines in Canada;
2. Underwater disposal in lakes or coastal waters;	Some environmental concerns over lack of containment and ability to control impact if necessary, Greatly reduced rate of oxidation for acid-generating tailings, Numerous examples in Canada but use is dependent on site-specific conditions;
3. Use as mine backfill.	Generally practical for only the coarse fraction of non-acid-producing tailings, Dependent on mining method, Used extensively in Canada.

For acid-generating tailings, underwater disposal prevents the oxidation of sulphides in the tailings. Few data concerning the long-term effect of the tailings on the quality of the body of water used for disposal exist currently. For this reason the regulatory agencies do not generally favour underwater disposal of tailings. It is being examined on a more general basis as part of the Reactive Acid Tailings (RATS)

Program. Nevertheless, this approach certainly warrants a site-specific evaluation, if a suitable body of water is available.

Underwater disposal is, or has been, used at the following mines in Canada:

Mine	Mine
Utah Copper, Rupert Inlet, B.C.	Wesfrob Mine, B.C.
Amax Kitsault Mine, B.C.	Cominco, Benson Lake, B.C.
Wabush Mines, Labrador	HBMS, Anderson Lake, Manitoba
Iron Ore Co. of Canada, Labrador	HBMS, Mandy Mine, Manitoba

At a number of abandoned mines in Manitoba acid-generating tailings were deposited into lakes. These situations are currently being investigated to determine both the long-term impact of the tailings on the lake and the chemical processes that have occurred in the tailings over time.

Two other methods of tailings disposal that are seeing some use at Canadian mines are the layered tailings (sub-aerial) and the thickened tailings methods. In the layered tailings disposal method, the tailings are deposited in thin uniform layers. Each layer, when completed is left to dewater and consolidate before another layer is added. In the second method, thickened tailings are discharged from a central spigotting point to produce a cone configuration. Both methods are discussed in Section 5.1.4.

Tailings are used for mine backfill at many mines in Canada, including:

Mine	Mine
Cominco	Ecstall
Falconbridge Nickel Mines	Westmin
Inco	Campbell Red Lake
Hudson Bay Mining & Smelting	Dome Mines
Kerr Addison	Falconbridge Copper
Noranda Mines	Giant Yellowknife Mines
Sherritt Gordon	Northgate Mines
Sigma Mines	Pamour Mines

In general, only the coarser sand fraction of the tailings is used because the backfilled material must be free-draining and have a low compressibility. The sands are

normally separated by cycloning so that only the slime fraction goes to a tailings impoundment area for disposal. Cement is sometimes added to the sand fraction to provide a free-standing, structurally competent backfill.

5.1.1 Tailings Impoundment Areas - General Characteristics. The discharge of tailings as a slurry into an impoundment area continues to be the most common method of tailings disposal in Canada. Tailings impoundments are economically attractive and relatively easy to operate. They can be designed to perform a variety of treatment functions effectively, including:

- removal of suspended solids by sedimentation;
- precipitation of heavy metals as hydroxides;
- perpetual retention of settled tailings and precipitates;
- flow balancing of storm water flows;
- equalization of wastewater quality;
- stabilization of oxidizable constituents such as thiosalts, cyanides, and flotation reagents; and
- storage and stabilization of process recycle water.

There are several disadvantages associated with tailings ponds:

- maintaining consistent control over the various treatment functions is sometimes difficult;
- good flow distribution is sometimes difficult to achieve;
- segregation of drainage from non-contaminated areas is often difficult;
- the large surface area associated with most tailings areas can pose reclamation problems, particularly in the case of acid-generating tailings;
- they are subject to wide seasonal variations in treatment performance, particularly with respect to their bio-oxidation efficiency;
- seepage through impoundment structures often requires collection and treatment; and
- dusting problems can be severe, unless surface stabilization is achieved by revegetation, use of chemical binders or rock cover.

The following sections review site selection, design, operation, and reclamation of tailings impoundment areas.

5.1.2 Tailings Impoundment - Site Selection. The following steps are recommended for the selection of a tailings impoundment site:

(1) Define volume and characteristics of tailings.

(2) Determine whether or not the tailings will be acid-producing by conducting the tests described in Section 3.1.4 to samples of the ore or tailings obtained through metallurgical test work.

(3) Define what functions the disposal area will need to perform in addition to the storage of tailings (see Section 5.1.1).

(4) Review disposal options listed in Table 11. If tailings impoundment is the preferred option, proceed with Steps (5) through (8).

(5) Review aerial photographs and large-scale maps to define alternative potential sites favouring the following characteristics:

- adequate capacity to store projected quantity of tailings for the life of the mine, preferably with potential for expansion in the event of increased ore reserves being found;
- adequate volume to provide water treatment functions, if applicable;
- proximity to mill;
- location in same watershed as mill, preferably down-gradient to minimize or eliminate pumping and provide a fail-safe system;
- minimum catchment area and ease of surface water diversion;
- area of impermeable overburden, non-recharge area for groundwater;
- suitable geotechnical conditions;
- availability of construction materials for retaining structures;
- ease of seepage control; and
- remote from public in case of radioactive tailings, otherwise unobtrusive.

(6) Undertake site reconnaissance or testing to define characteristics further, and rank potential sites.

(7) Undertake pathway analysis in the case of radioactive tailings disposal sites.

(8) Undertake detailed technical, environmental, and economic evaluation of preferred site (or sites if the ranking is close), including implications of reclamation.

5.1.3 Tailings Impoundment - Design and Development. The design and development of a tailings impoundment for a specific mine/mill complex depends largely on the following:

- potential impact of the impoundment on surface and groundwater. Local hydrogeological conditions must be understood and the potential impact of leachate from the tailings evaluated. (A model to evaluate radioactive pathways is being developed for tailings containing radionuclides through the National Uranium Tailings Program. Similar predictive techniques may ultimately be available for acid-generating tailings through the Reactive Acid Tailings Program.) Depending on

the evaluation of the in-situ conditions, some form of liner may have to be considered for the impoundment area;

- type of retaining structure to be used;
- hydrologic characteristics of the site, including flood routing capacity, type of spillway or decant;
- method of operation, including sub-aqueous or sub-aerial deposition, use of tailings in the retaining structure, location of pool relative to dam, and recycle requirements;
- final configuration required for reclamation, including flooded or dry, surface stabilization and drainage, seepage control, vegetative or other covers;

All tailings containment structures should be designed and constructed in accordance with accepted engineering practices and in compliance with the geotechnical characteristics of the site, the dam construction materials, local seismic and hydrological conditions, as well as the physical configuration of the structure in the context of the local topography.

Although the same basic design principles apply, tailings storage systems differ from conventional water-retaining dams in the following ways:

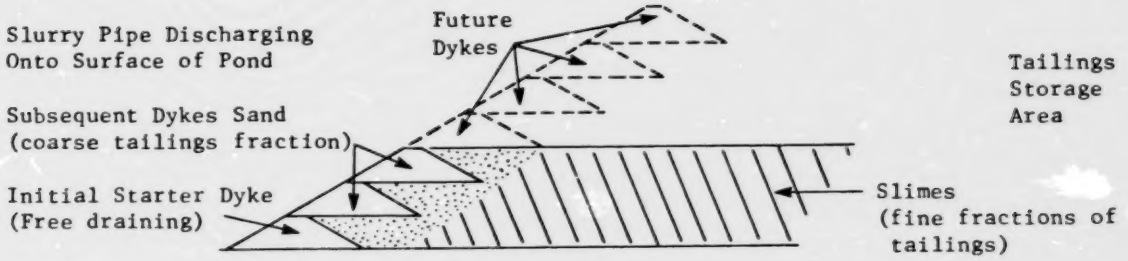
- the material stored often consists of slime-sized solids;
- the dam is raised frequently as further storage volume is required;
- the coarse fraction of the tailings is often used in the construction of the dam; and
- the stored tailings may be chemically unstable, or may, in the case of uranium tailings, produce radon and other decay isotopes of uranium. With unbuffered sulphide tailings, sulphuric acid and associated leaching products will be produced.

The following types of retaining dams are commonly used for tailings impoundments:

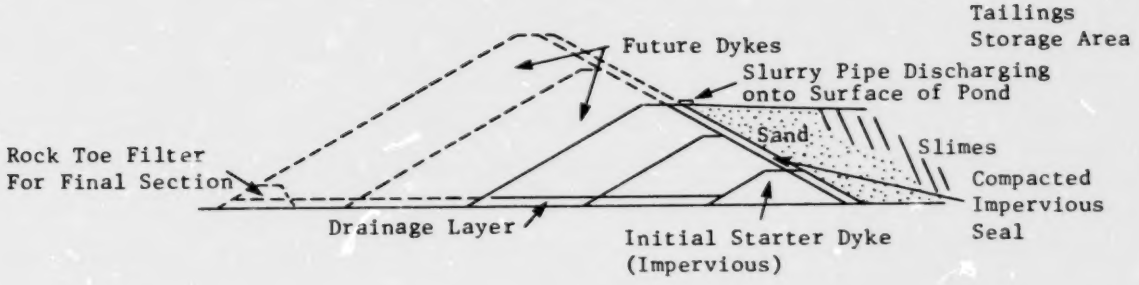
- borrowed material dams;
- dams constructed of tailings using 'upstream' method;
- dams constructed of tailings using 'downstream' method;
- dams constructed of tailings using 'centreline' method; and
- combinations of the above.

The basic characteristics of the three methods using tailings as the primary construction material are shown in Figure 2. All types are briefly reviewed in the following text.

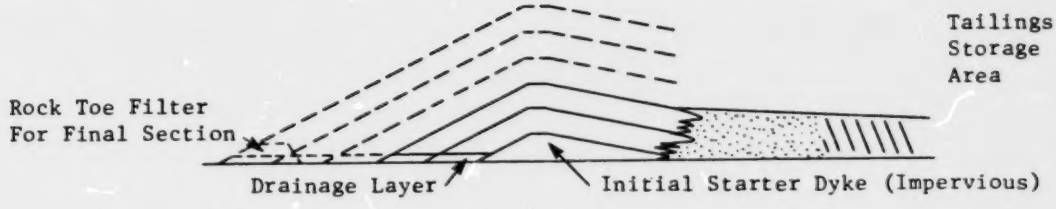
Borrowed material dams have an advantage in that the quality of the materials and their placement can be accurately predicted and controlled. There is an increasing tendency in Canada and elsewhere to use zoned dams in which all or some of the material



UPSTREAM METHOD



DOWNSTREAM METHOD



CENTRELINE METHOD

FIGURE 2 TAILINGS DAM CONSTRUCTION METHODS

is borrowed. This is particularly true with retaining structures for uranium tailings and acid-producing sulphide tailings, since these materials should not be used in the exposed downstream sections of an impoundment structure. Waste rock that is potentially acid-generating should not be used in exposed sections of impoundment structures either.

The principal disadvantage of using borrowed materials is the cost of excavation and placement relative to the use of tailings, plus the fact that sufficient borrowed material may not be available at some sites.

Upstream method of dam construction using cycloned or spigotted tailings was used extensively in the past but not now because of the inherent difficulty of ensuring its stability under seismic loadings. As indicated in Figure 2, a typical upstream section incorporates the finely divided slimes fraction of the tailings into the structure and can therefore be susceptible to failure through liquefaction.

Downstream method of dam construction provides the opportunity to design and construct a dam to meet acceptable geotechnical specifications. For it, the coarse fraction of tailings is used.

The major disadvantage of the downstream method of construction is the large volume of sands required to raise the dam. In the early stages of operation, or where the ore is relatively finely ground, it may not be possible to produce a sufficient volume of sands to maintain the crest of the tailings dam above the rising pond level, particularly when the quantity of material required for a unit increase in height increases as the structure progresses. Most downstream construction applications use some variation of cycloning to segregate the coarse tailings and divert them to the outer face of the dam.

Centreline method of dam construction maintains the dam centreline in the same vertical plane as the height increases. It can be regarded as a compromise between the upstream and downstream methods and can provide an acceptable structure where the classification of upstream tailings can be controlled to ensure their stability as an integral part of the structure under prevailing seismic and geotechnical conditions. Tailings should not be used as a dam construction material when they are either radioactive or acid-producing because of the exposure of these materials on the downstream face.

Most tailings impoundments produce seepage flows at the downstream toe of the retaining structure. Where the quality of this seepage exceeds the effluent requirements (see Section 2), it is common practice to construct a downstream seepage collection system and treat the water collected prior to discharge (see Section 5.3).

Spillways, decants and diversions. The most common causes of failure associated with tailings impoundment structures are those associated with spillways,

decants, and diversions. If the hydraulic capacity of these facilities is exceeded during periods of peak flow, a failure of the impoundment structure itself can result, leading to the liquefaction and release of the stored tailings.

All tailings impoundments must be thoroughly evaluated at the design stage to ensure that rainfall/runoff events, corresponding to accepted statistical return criteria for intensity and duration, can be accommodated through storage, diversion around the facility, or discharge via a spillway or decant structure.

Determination of the levels of design flood requires a statistical analysis of hydrologic data available for the area. The analysis must take into account the fact that tailings impoundments often control relatively small drainage areas with little storm attenuation capability.

Where potential failure of a tailings system has life-threatening implications, and particularly when a downstream community might be involved, a one in 10 000 year or even greater probable runoff event should be used in establishing the design criteria. This should take into account the combination of rainfall events and snowmelt as applicable to the specific area. Design criteria should be reviewed and agreed upon with the appropriate regulatory authorities. (Refer to the International Conference on Large Dams (ICOLD) and United States Bureau of Reclamation (USBR) standards for dam design as well as to the National Research Council publication "Flood Hydrology in Canada - A Summary of Current Practice".)

Unlike water storage dams, tailings impoundment systems usually do not include an overspill section. Such a structure is difficult to maintain because a tailings impoundment is raised throughout the life of the mine. A more common approach is the use of decant towers, which can be raised as the depth of tailings increases. Failure of these systems due to settlement, seismic loadings, or blockages has been a problem in the past, and it is important that the design of decant systems fully anticipates the possibility of such events. Some form of back-up system should be provided in case problems with the primary decant arise.

Other forms of releasing excess supernatant from tailings impoundments include siphon systems and barge-mounted pumps.

Siphons have an important advantage in that they can operate at full capacity for a relatively small rise in the level of the pond.

Disadvantages include:

- inability to pass ice and debris;

- possibility of water freezing in the inlet legs and air vents before the pond rises sufficiently to prime the siphon;
- occurrence of surges and stoppages as a result of erratic make-and-break actions of the siphon;
- susceptibility to cavitation and low absolute pressures (for this reason, they are usually limited to a total drop of 7 m on earth dams); and
- possibility of structural failure of the pipes due to settlement of the embankment.

Barge-mounted pumps are a commonly used method of decanting a tailings area, particularly where all or most of the water is used for recycle to the mill. They have the advantage of being flexible with respect to location; that is the tailings impoundment can be operated with the pond remote from the dam (see Section 5.1.4). Tailings systems using pumps must be designed so that peak runoff events can be safely stored within the impoundment and subsequently released.

5.1.4 Tailings Impoundment - Operation. Many factors must be considered in selecting the method of operating a tailings pond. These include: method of depositing tailings, size and location of the settling pond, entry and decant arrangements, final configuration required for reclamation, treatment functions to be performed by the pond, and the use of tailings for dam construction.

Method of tailings disposal. Tailings can be disposed of by various methods, some conventional, others relatively new.

Conventional methods of tailing disposal. Most mills pump their tailings with a solids content between 15 and 55 percent by weight, usually averaging about 40 percent solids to a disposal area. Pipeline velocities of 1.6 to 3.0 m/s are used depending on the coarseness and size distribution of the solids, pulp density of the slurry, and other factors. Pipe wear is an important design factor and either rubber-lined steel pipe or high density polyethylene pipe are normally specified to counter this problem.

Where the tailings impoundment is down-gradient of the mill, gravity launders or pipelines are used if topography permits. Concrete drop structures are used to maintain the pipeline grade at 1 percent or less in order to prevent excessive abrasion and extreme pipe pressures. Provision must be made for pipe drainage in the event of a mill shutdown, so that tailings do not settle in the line and plug it.

Tailings can be discharged into a large pond area (sub-aqueous deposition) or onto a tailings beach removed from the pond (sub-aerial deposition). The latter approach can involve discharge from a single point, through a series of spigots from a header,

through spray bars, or through cyclones for the mechanical separation of tailings sands for dam construction.

Where tailings are spigotted from the crest of a retaining dam, the coarse material deposited close to the point of discharge can provide a source of material for raising the dam. This approach also results in the tailings pond being remote from the dam, which has the advantage of lowering the phreatic surface at the dam and thus reducing seepage volumes. The final configuration of the tailings deposited in this way is that of a large dish, sloped inwards toward the final pond area or discharge point. There may be difficulties associated with the abandonment of the impoundment area in this configuration in terms of by-passing runoff or maintaining an acceptable quality of discharge, particularly if the tailings are radioactive or acid-producing.

Discharge of the tailings through cyclones has the advantage of providing a greater degree of control over the size distribution of sands used for dam building. Large volumes of coarse tailings are required for both the centreline and downstream methods. Close control over permeability must be exercised to ensure adequate drainage of the impoundment structure. Cyclones are usually mounted on the dam crest so that the location of the coarse and fine discharges can be controlled. This method is generally not applicable to finely ground tailings because the yield of coarse material for dam construction is inadequate.

Sub-aqueous method of tailings deposition. If tailings are potentially acid-producing, the use of sub-aerial deposition promotes the oxidation of the tailings on the exposed beaches where the sulphide material is continuously exposed to air and moisture from rainfall and capillary action. Heating of the beach areas by direct sunlight in summer further enhances the oxidation reaction. For this reason the use of sub-aqueous deposition into a large pond covering the deposited tailings may prove a preferable alternative for chemically reactive tailings, particularly if perpetual flooding is a feasible reclamation option (see Section 5.1.5).

Layered method of tailings disposal. With the layered method spray bars are used to deposit tailings slurry in thin layers of uniform thickness (100 to 150 mm). The slope of the deposited layer depends on the characteristics of the slurry and usually varies between 0.5 and 1.0 percent. This method is often referred to as "sub-aerial" tailings deposition.

Once a section of beach has been covered with the desired thickness of fresh tailings, discharge is started on another section of beach and the newly deposited layer is left to settle and drain. The time required for settling and drainage ranges from several

hours to a few days. The drained section is then left to dry. Evaporation produces a negative pore pressure within the layer of tailings that results in further consolidation. The percentage of solids at the time of placement influences the final degree of consolidation. The final configuration is a gently sloping mass of tailings, made from uniform layers, with the resulting vertical permeability several orders of magnitude lower than horizontal permeability.

This method has been applied to about 40 tailings area around the world in the last 25 years. In Canada, it is used at the Key Lake uranium mine, in Northern Saskatchewan and at the Westmin copper-zinc mine on Vancouver Island. At Key Lake, during the winter months, conventional tailings discharge is being used, at a different disposal site than that used for layered tailings disposal.

The layered tailings disposal method supposedly offers the following advantages (Knight and Haile, 1983):

- low vertical permeability of the tailings mass (10^{-7} to 10^{-9} cm/s);
- high run-off coefficient (0.85 to 0.98);
- at decommissioning, the tailings are well drained and fully consolidated, thus eliminating long-term seepage; and
- relatively solids-free supernatant because of the low velocity of water flow over the tailings surface.

The main disadvantages of the methods are:

- its high initial capital cost due to site preparation; in Canada, an underdrainage system is probably necessary for the method to be effective (Key Lake);
- it cannot be used year-round under cold climate; and
- it is still unproven at Canadian mines.

Thickened tailings disposal. The second relatively new tailings disposal method is called the thickened tailings disposal. In this case, tailings slurry is thickened prior to its discharge from one or more spigotting points within the tailings disposal area. The tailings form a cone with a slope ranging between 2 and 8 percent. The slope depends on such factors as the solids content, fineness, and specific gravity. Typical percent solids required to achieve a slope of 6 percent range from 54 to 74 percent. Tailings of such consistency flow as a slurry, with no size segregation occurring.

This method has the following advantages (Robinsky, 1975):

- greater tailings storage within a given disposal area than with conventional tailings disposal methods;

- no requirement for large dams associated with many conventional disposal schemes;
- minimized dusting during and after operations;
- retention of fine particles within the tailings mass;
- prevention of downward seepage through the tailings mass by capillary rise potential;
- low seepage through retaining structures;
- from the same spigot that was used for tailings discharge, entire tailings surface may be easily covered with inert material, top soil or fertilizer for reclamation purposes.

Some of the disadvantages of the method include:

- higher operational cost than conventional methods because of the need to thicken the slurry;
- possible operational problems with thickener operation; and
- higher erosion rate of the tailings from precipation during operations, resulting in a higher solids content in the supernatant pond.

The thickened tailings disposal method is currently being used at the Kidd Creek mine in Ontario and at the Selbaie mine in Quebec.

Size and location of settling pond. An important requirement for efficient operation of a tailings impoundment is that the pond is of sufficient size to ensure adequate settling and clarification of the decant (or recycle water) prior to its discharge from the pond.

The size of the pond required for satisfactory clarification of solids is determined by a number of considerations, including particle size, tendency to slime (clay type minerals), pH and temperature of the water, quiescence of the pool (i.e., lack of wave action), depth of water, and inlet and outlet configurations.

The following guidelines can be used as an aid to the design and operation of tailings ponds, although actual requirements are very specific to both the characteristics of the waste and the local climatic conditions:

- the pool should provide at least a 5-day retention time for normal clarification and be designed and operated to eliminate short-circuiting;
- the area of the pool should be sized to provide 5 to 12 hectares for each 1000 tonnes of tailings solids discharged a day. An average of 7 hectares per 1000 tonnes a day is usually considered adequate under normal conditions;
- where chemical or biochemical stabilization is a treatment function of the pond, a minimum retention of 30 days should be provided. Long-term storage may be required during the winter months when natural stabilization processes may have little effect;

- a minimum depth of 2 m should be maintained at the point of discharge to reduce the impact of wind and wave action on sedimentation; and
- the amount of freeboard provided above the maximum pond level should be selected on the basis of the fetch provided by the pond to prevailing winds. The freeboard is normally about 1 m.

The implications of the pond configuration within a tailings impoundment are illustrated in Figures 3 and 4.

5.1.5 Tailings Impoundment - Reclamation. The approach to reclaiming a tailings impoundment must be considered an integral function of the site selection, system design and operation.

Rehabilitation issues to be considered and resolved at the pre-development stage include:

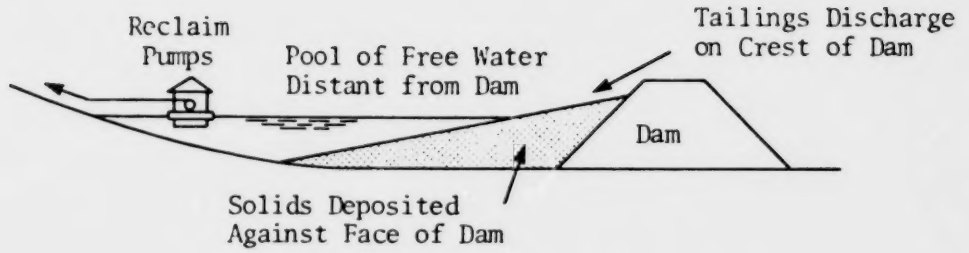
- long-term geotechnical integrity of the impoundment;
- chemical stability of the tailings, particularly whether they are acid-generating;
- post-operational surface and groundwater regimes including surface water routing, hydrogeological characteristics, need for seepage collection, long-term spillway requirements, etc.;
- sources of material for surface cover/stabilization;
- dry or flooded abandonment system; and
- final surface topography of the impounded tailings, if dry.

These factors are addressed in Section 6.5 under post-operational considerations.

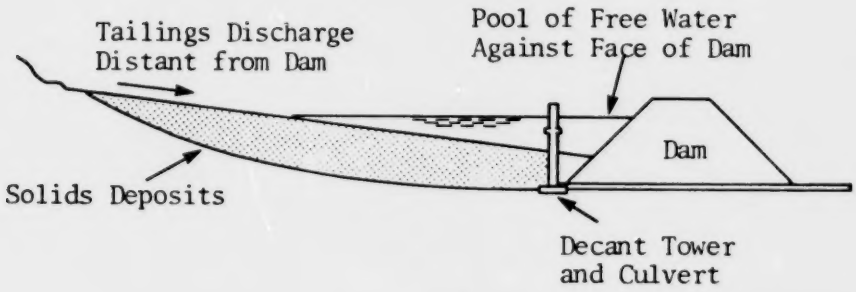
5.2 Mine Water

The volume and quality of mine water vary widely depending on the type of mining operation, geochemistry, and hydrology/hydrogeology of the specific mine. For open-pit operations, the volume of water depends on the ingress of groundwater and the precipitation onto the surface area of the pit. The water is normally collected in sumps at the bottom of the pit and pumped to the surface for treatment, use in the mill, or discharge. Water from open pits may contain traces of nitrogen-derived residuals from blasting operations and normally require sedimentation for the removal of suspended solids. This may be provided within the sumps in the pit.

Acidic pit waters must be neutralized and have their metals removed. This can be achieved in a treatment system specific to the mine drainage or, more normally, in the tailings pond or other collective system also treating contaminated surface drainage.



DISCHARGE ON DAM



DISCHARGE DISTANT FROM DAM

FIGURE 3 EFFECTS OF DISCHARGE LOCATION

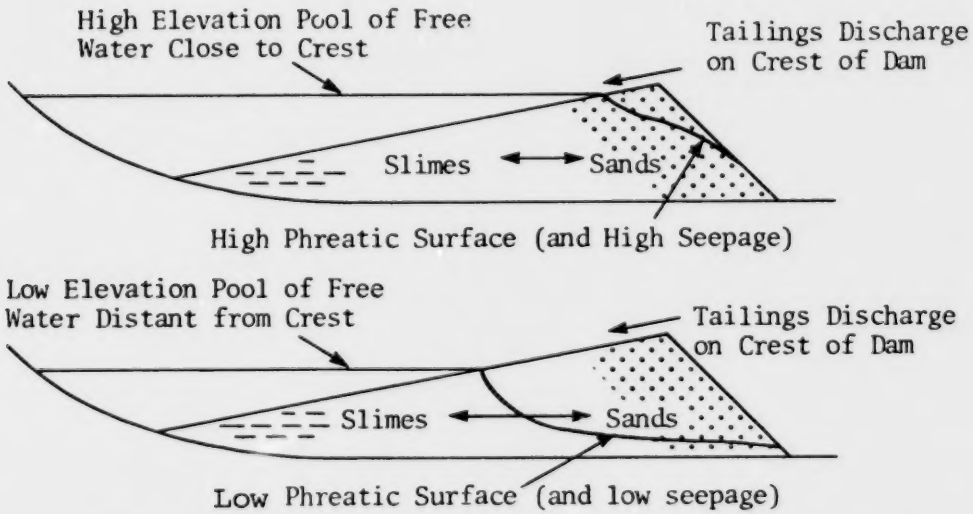


FIGURE 4 EFFECTS OF POND LEVEL ON SEEPAGE CHARACTERISTICS

For underground operations the volume of mine water is influenced by the local hydrogeologic conditions, the infiltration of surface waters into the mine, water used for drilling and dust suppression, and water contained in tailings backfill where this is used. Mine water is pumped to the surface from collection sumps and, depending on its quality, used as process make-up, treated, or discharged directly.

An indication of the variability in mine water volume and quality at a particular operation is provided in Table 12 compiled from data from Cominco's Sullivan mine, an underground operation of 6500 tonnes/day.

TABLE 12 MINE WATER: VARIABILITY IN VOLUME AND QUALITY

Parameter	Range
Volume (L/min)	1200-10 000
Fe (mg/L)	200-1000
Zn (mg/L)	30-200
Mn (mg/L)	20-75
Al (mg/L)	35-150
Pb (mg/L)	2-5
S.S. (mg/L)	200-900
pH	2.8-4.0
Temperature (°C)	8-12

5.3 Surface Drainage and Contaminated Seepages

The factor deciding whether surface drainage is a significant concern is the acid-generating potential of exposed mineralization, waste rock, and tailings. The acid generation tests described in Section 3.1.4 must be undertaken, at the pre-development stage for all new mines yielding sulphidic ores, so that appropriate measures for minimizing and controlling acid drainage at mines can be incorporated into the initial design of the operation. The precautions for dealing with acid surface drainage include:

1. Avoiding or minimizing the exposure of sulphide-bearing minerals where possible.
2. Avoiding the use of sulphide-bearing material for the construction of dams, roads, or other fill requirements.
3. Placing acid-producing waste rock in areas where seepages can be controlled.
4. Sealing all faces of waste rock piles with impermeable material on a regular basis to reduce the access of air and water.
5. Segregating clean and contaminated drainage to the maximum degree feasible.

6. Designing and constructing drainage control facilities to accepted criteria for surface runoff intensity and duration.
7. Designing surface drainage facilities as an integral part of both the operational and post-operational control systems.

Acid drainage control systems for surface drainage will normally incorporate balancing ponds to reduce peak flows and thus lower the capacity of the treatment system required.

For sites that are not acid-producing, the drainage facilities are normally relatively simple and primarily provide for the control of storm runoff from the site with a minimum of surface erosion. Depending on the topography and grades of a given site, sediment traps may be required to minimize the impact of suspended solids on local receiving systems. Stabilization of slopes and channels by revegetation, rip-rap, or other means are important considerations in the design and maintenance of surface drainage systems.

5.4 Mechanical Treatment Systems

During the period from 1978 to 1986, the mining industry in Canada brought on line approximately 25 mechanical treatment systems designed to provide neutralization or the removal of metals and other substances from process effluents and mine drainages. Before this time, virtually all treatment at Canadian mines was provided by pond-based systems.

Canadian mines using these mechanical wastewater treatment systems are:

Base metal mines	Gold mines	Iron ore mines
Westmin	Giant Yellowknife	Mount Wright
Sullivan	Con	
Waite Amulet	Equity Silver	
Geco	Mount Skukum	Uranium mines
Mattabi	McBean	
Inco Copper Cliff (2)	Golden Giant	Amok
Falconbridge	Tech Corona	Key Lake
Gallen	Kiena	Eldor
Brunswick Mining (2)	Muschocho	Rio Algom Stanleigh

This section of the report provides flow diagrams and descriptions to illustrate the various types of mechanical wastewater treatment systems in operation at Canadian mines.

5.4.1 Treatment of Acid Mine water. The flow diagram for treatment of the metal-containing acid mine water, generated in the underground workings at Brunswick Mining's operation in northeastern New Brunswick, is given in Figure 5 (Wasserlauf, 1985).

Mine water flows by gravity from a balancing tank into a neutralization tank where lime slurry (18 to 20 wt. percent solids) and air are added. Air is added to oxidize ferrous iron. The first neutralization tank overflows into a second tank which provides additional residence time for precipitation of the metal hydroxides and gypsum. The contents of the tank are sparged with air to ensure the complete oxidation of ferrous iron. The overflow from this tank discharges into the mill's tailings pond system for solids-liquid separation. The resultant sludge is retained in the tailings pond for permanent disposal.

The plant is operated at a pre-set flow rate of influent. The addition of lime slurry is automatically controlled by a pH control system.

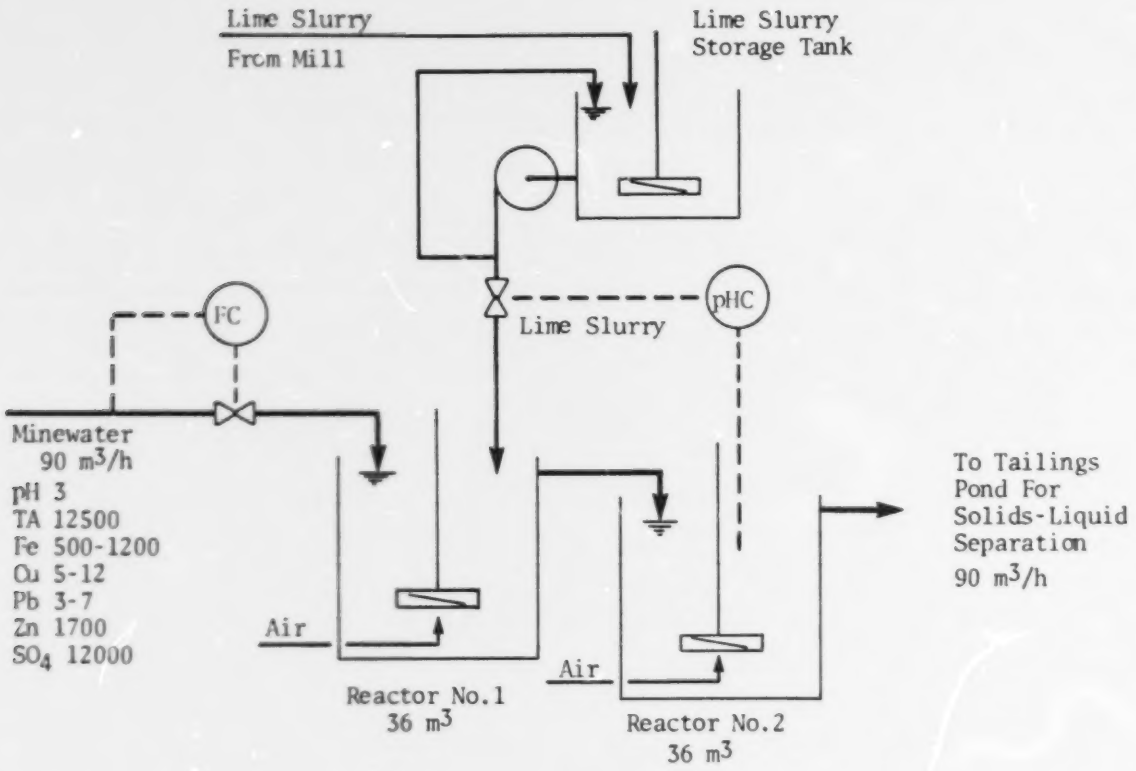
The plant, as originally designed and operated, included a polymer flocculant addition system and a plate-type clarifier. However, continuous difficulties with poor sludge settling and pumping of the sludge led to these systems being discarded. Currently, the neutralized effluent from the treatment system is discharged into the tailings pond.

The formation of a very hard gypsum scale on treatment plant equipment is a continuing problem. Prevention or minimization of scale formation by the use of antiscalants and linings has not been successful. At present, scale is periodically removed using high pressure water jets and manual labour.

The capital cost of the system as originally constructed was approximately $800 000 (Cdn., 1985). Annual operating costs are approximately $550 000, of which $500 000 is the cost of reagents.

5.4.2 High-Density Sludge Process for Treatment of Acidic Wastewaters. The flow diagram for the treatment of acidic seepages at Noranda's inactive Waite Amulet property in Quebec is shown in Figure 6 (Wasserlauf, 1985).

Acidic water flows from a holding pond into a neutralization tank where it contacts a mixture of lime and previously precipitated sludge. Air is sparged into the tank to oxidize ferrous iron. The overflow from this tank discharges into a second neutralization tank which provides additional residence time for the precipitation of metal hydroxides and gypsum as well as the oxidation of ferrous iron. The overflow from the second neutralization tank discharges into a plate-type clarifier for solids-liquid separation. A solution of polymer flocculant is metered into the neutralization tank



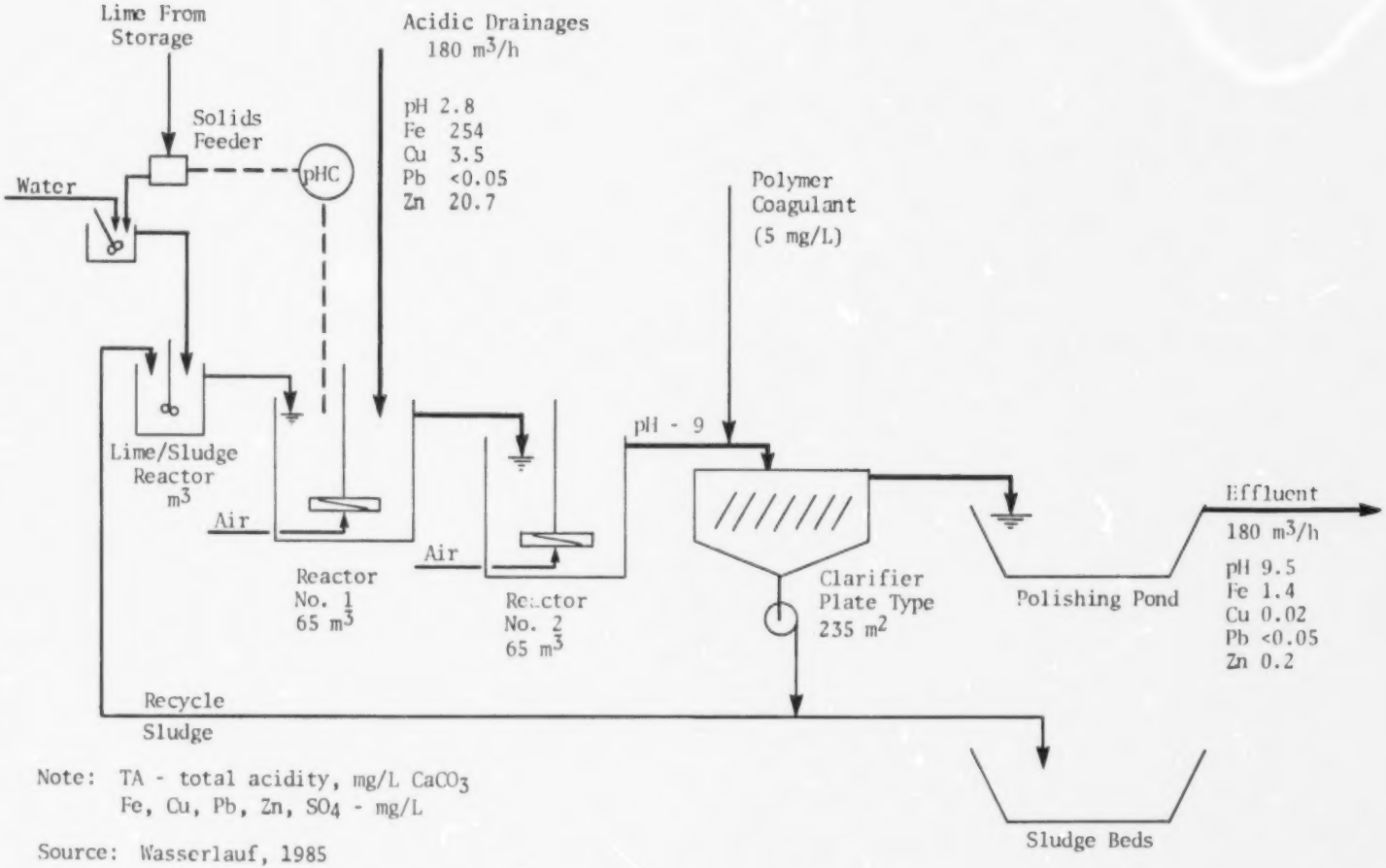
Note: TA - total acidity, mg/L $CaCO_3$
Fe, Cu, Pb, Zn, SO_4 - mg/L

Source: Wasserlauf, 1985

FIGURE 5 ACID MINE WATER TREATMENT SYSTEM

discharge launder upstream of the clarifier. Overflow from the clarifier discharges into a polishing pond. Sludge from the clarifier (about 4 percent (wt.) solids) is permanently disposed of in sludge drainage beds (underlain by sands).

5.4.3 Treatment for Cyanide and Arsenic Removal. Various processes have been and are being used for cyanide removal. Initially only the barren bleed solutions were treated, but because washing the tailings filter cake sufficiently free of cyanide is difficult, and also, the need to cope with unfiltered tailings slurries from CIP plants, more and more tailings pond overflows instead are being treated. In this way the solids are settled out.



Note: TA - total acidity, mg/L $CaCO_3$
Fe, Cu, Pb, Zn, SO_4 - mg/L

Source: Wasserlauf, 1985

FIGURE 6 ACID MINE WATER TREATMENT HIGH-DENSITY SLUDGE PROCESS

Furthermore, if the retention time in the pond is long enough, there is a significant reduction in cyanide concentration due to natural degradation. One or two mills in Canada also treat CIP barren slurry directly for cyanide destruction.

Where the wastewater contains arsenic, an arsenic removal step is added, usually subsequent to one of the oxidative cyanide destruction processes, such as alkaline chlorination, hydrogen peroxide or Inco SO_2-air. In this way, any arsenite ion is oxidized to arsenate during cyanide treatment. A solution of an iron salt, usually ferric sulphate, is then introduced to precipitate basic ferric arsenate.

The following plant descriptions are examples of the application of some of the more commonly used processes to treat both waste barren solution and tailings pond overflows. Summaries of operating data illustrating cyanide removal efficiency and reagent consumptions, as well as capital and operating costs where available, are also given.

The process plant descriptions illustrate:

- cyanide and arsenic removal - Giant Yellowknife Mines.
- cyanide removal - McBean Mine.
- cyanide and antimony removal - Noranda Golden Giant Mine.
- cyanide and arsenic removal - Con Mine and Teck-Corona Mine.

Giant Yellowknife Mine. The Giant Yellowknife mine treats an ore in which gold values are locked within grains of arsenopyrite. This necessitates preconcentration followed by roasting of the resulting sulphide concentrate. As a result of the roasting step, the leach feed does not contain labile iron sulphides, and hence ferrocyanide is absent. Arsenic is present as arsenite, a form which does not yield insoluble compounds. Thus the chlorination process serves not only to destroy cyanide but also to oxidize the arsenite to arsenate, permitting its subsequent precipitation as a basic ferric arsenate compound. A sizable fraction of the chlorine consumption should therefore be attributed to arsenic removal rather than to cyanide destruction.

Operating Parameters: (1984 operating season)

Solution feed rate:	8 890 m^3/day (May to October only)
Total volume treated:	1 554 720 m^3 (175 days).

The treatment plant, which is operated only during the ice-free period (May to October) comprises three stages (see Figure 7).

First stage: cyanide and metals are removed by lime and chlorine addition,

Second stage: arsenic reacts with added ferric sulphate,

Third stage: the precipitate produced is treated with flocculant to yield a settleable product. The treated effluent is discharged to a large polishing pond with a 15-day retention.

(See Table 13 for performance data, Table 14 for cost data, and Table 15 for a cost breakdown for cyanide and arsenic removal.)

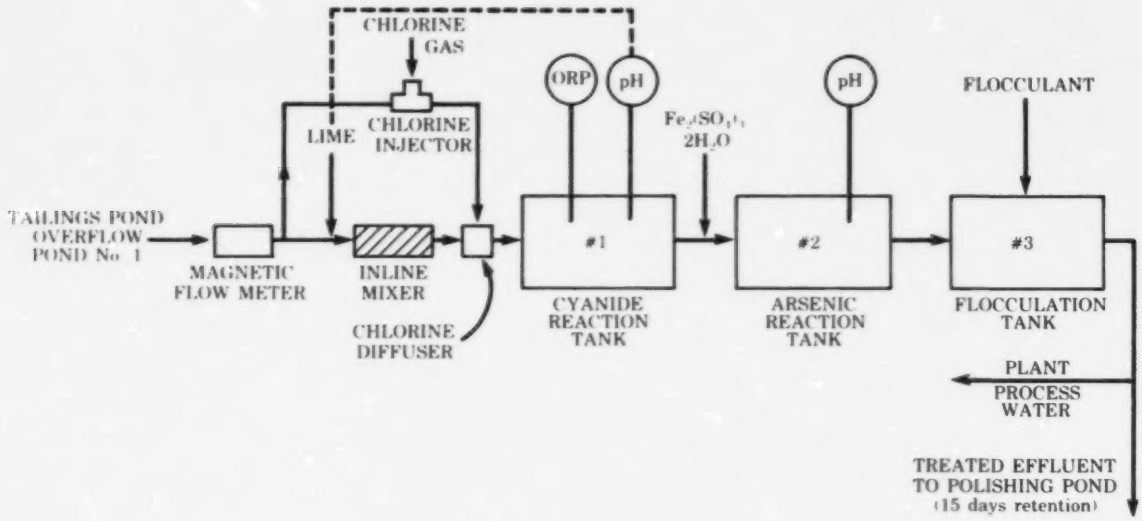


FIGURE 7 GIANT YELLOWKNIFE MINE - EFFLUENT TREATMENT PROCESS

TABLE 13 PERFORMANCE DATA (GIANT YELLOWKNIFE MINE)

	Constituent (mg/L)					
	CN_{WAD}*	CN_T**	Cu	Fe	Zn	As
Influent	7.1	7.5	6.7	<0.1	0.1	12.1
Polishing Pond Overflow	0.09	0.15	0.03	<0.1	0.1	0.14
% Removal	98.7	98.0	99.6	-	0	98.8

* CN_{WAD} - cyanide weak acid dissociable
** CN_T - total cyanide

TABLE 14 COST DATA (GIANT YELLOWKNIFE MINE)

Capital Cost of Plant	$2 000 000 (Cdn., 1981)
Operating Costs	
Material*	$ 569 000
Labour	$ 82 000
Total	$651 000

* Amount of Reagents Consumed:

Lime	505	tonnes
Chlorine	218	tonnes
Ferric sulphate	700	tonnes
Flocculant	4.3	tonnes

TABLE 15 COST BREAKDOWN (ESTIMATED) FOR CYANIDE AND ARSENIC REMOVAL (GIANT YELLOWKNIFE MINE)

	Cyanide	Arsenic
Amount removed (t)	12.0	20.7
Cost/kg ($)	23.70	16.40
Cost/tonne of ore* ($)	0.77	0.91

* Amount of ore milled: 386 250 tonnes

McBean Mine. The McBean Mine treats 600 tonnes of ore a day. The ore contains both pyrite and pyrrhotite minerals, and cyanide consumption averages 3.5 kg/tonne (see Figure 8).

The effluent treatment system treats barren bleed only, and consists of a 4-compartment flotation cell having a total volume of 20 m^3. The first compartment has a capacity of 2 m^3, each of the other three 6 m^3. Copper sulphate solution is fed to the first compartment and sodium bisulphite solution and lime slurry (for pH control) are fed to the second. This gives a nominal retention time of 30 minutes at the normal flow of 200 L/min, which provides sufficient time to complete the cyanide removal reactions. The pH is controlled at 8.5; air is supplied by a blower at 1 L/min/L reactor volume. The air, bisulphite and influent flows, as well as the pH, are kept constant. Treatment performance is monitored by observing the colour of the treated wastewater in the

reactor and by conducting cyanide analyses. (See Table 16 for performance data, Table 17 for cost data, and Table 18 for a cost breakdown of cyanide removal.)

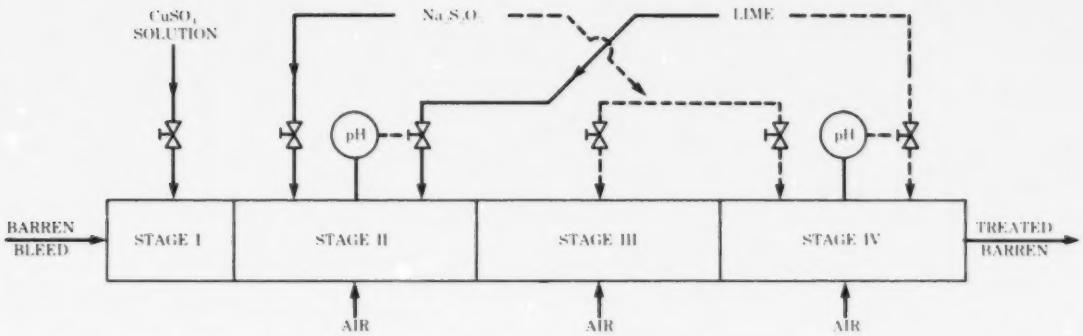


FIGURE 8 McBEAN MINE - EFFLUENT TREATMENT PROCESS

TABLE 16 PERFORMANCE DATA (McBEAN MINE)

	Constituent, (mg/L)					
	CN_{WAD}	CN_T	Cu	Fe	Zn	As
Influent	225	366	35.5	45.8	62	4.1
Effluent	0.16	0.69	0.60	0.27	0.30	0.23
% removal	99.9	99.8	98.3	99.4	99.5	94.4

Noranda-Golden Giant Mine. The Golden Giant mill has a current capacity of 2300 tonnes per day and uses the CIP process. Tailings and mine water are directed to a tailings pond. The tailings pond supernatant becomes the feed for the treatment plant (see Figure 9). The treated effluent is discharged to Lim Lake. In addition to cyanide, copper, zinc, and nickel, the treatment plant feed contains antimony. The flow sheet incorporates a separate circuit, similar to the circuits used for arsenic removal.

TABLE 17 COST DATA (McBEAN MINE)

Capital Cost of Plant (estimated) (using new equipment; the McBean Plant was assembled from used components)	$167 000 (Cdn.)
Operating Costs	
Chemicals*	$ 42 525
Other supplies	$ 42 525
Labour**	$ 36 450
Royalty	
Total	$121 500

* Amount of reagents consumed:

Lime	32 tonnes
Sodium metabisulphite	63.75 tonnes
Copper sulphate	1.716 kilograms

** Labour constitutes 15% of the total treatment plant operating time; (i.e., 2169 h x 0.15 = 325 h).

TABLE 18 COST BREAKDOWN FOR CYANIDE REMOVAL (McBEAN MINE)

Amount of cyanide removed (t)	8.28
Volume of solution treated (m^3)	22 123
Cost/kg of cyanide removed ($)	14.63
Cost/tonne of ore* ($)	0.52

Cyanide Treatment	
SO_2 feed rate	500 mL/min, added to the first reactor
pH at reactor No. 1	8.5
Antimony Removal	
Fe to Sb ratio	10:1 (added as 100 g/L Fe)
Ferric sulphate solution feed rate	1.7 to 2.5 L/min
pH at reactor No. 1	7.8

* Amount of ore milled: 231 284 tonnes

Antimony is removed by reaction-coprecipitation using ferric sulphate and lime. The ferric sulphate source is waste pickle liquor containing 100 g/L iron. (See Table 19 for operating data and Table 20 for performance data.)

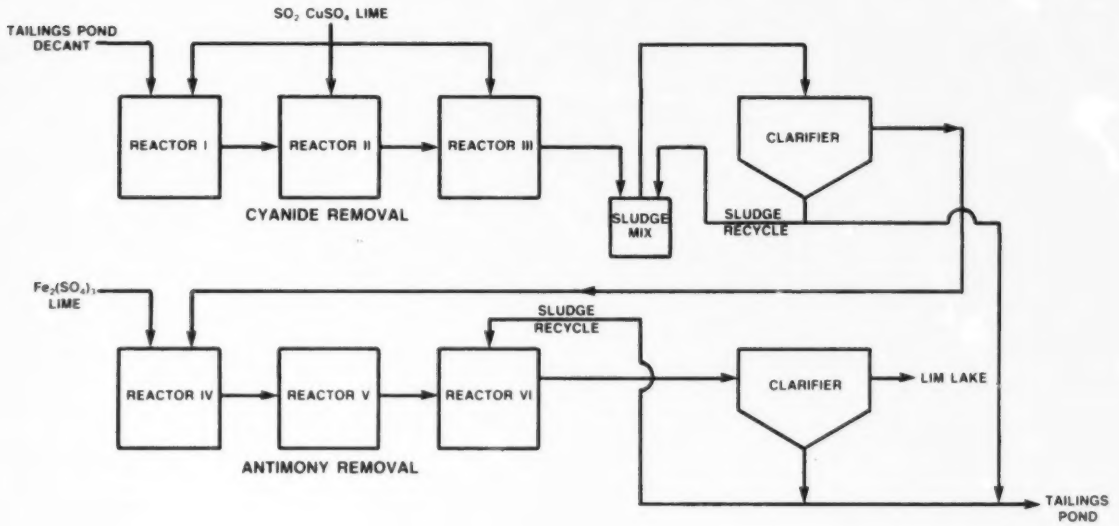


FIGURE 9 GOLDEN GIANT MINE - EFFLUENT TREATMENT PROCESS

TABLE 19 OPERATING DATA (GOLDEN GIANT MINE)

Influent flow rate	190 m^3/h
SO_2 to CN_T ratio	6-7:1 (100% liquid SO_2)
Cu to CN_T ratio	0.5:1 (15 mg/L solution)

TABLE 20 PERFORMANCE DATA (GOLDEN GIANT MINE)
(For test run, April 1986, using SO_2 and ferric sulphate)

	Constituent, (mg/L)			
	CN_T	Cu	Fe	Sb
Influent	50	5.5	5.1	11.2
Effluent	0.1	0.04	0.04	2.7
% Removal	99.8	99.3	99.2	75.9

Con Mine. The Con Mine treats 700 tonnes of ore per day by the Merrill Crowe process. It has used the hydrogen peroxide process for cyanide removal since February 1986 (see Figure 10). For many years the mill feed contained arsenopyrite which necessitated its roasting. The arsenic fumes were collected in scrubbers and pumped to an enclosed concrete pond. Roasting was discontinued in 1970; and in 1982, in response to the development of a market for arsenic, an arsenic trioxide recovery plant was added. The effluents from the recovery plant discharge to the same tailings area as the cyanidation tails. Thus, treatment is, like that at Giant Yellowknife, in two stages: the first to remove cyanide and copper, the second to remove arsenic. The first-stage oxidant, in this case hydrogen peroxide, must not only destroy cyanide, but must also oxidize arsenite to arsenate for subsequent precipitation with ferric iron in the second-stage reactor. Three reactors in series are used. The overflow from the tailings pond is pumped, along with copper sulphate as a catalyst, to the first reactor, where it is treated with hydrogen peroxide. It then overflows to the second reactor where ferric sulphate solution is added along with lime to maintain a pH between 7 and 8. The effluent from this reactor exits to a third tank to provide additional reaction time. The overflow, after a flocculant has been added, enters a clarifier. A large fraction of the clarifier underflow is recycled to the second tank, whereas the remainder is pumped to the tailings pond. The treated clarifier overflow is released to the environment. (See Table 21 for operating data and Table 22 for performance data.)

TABLE 21 OPERATING DATA (CON MINE)

Solution rate		272 m^3/h	
Reactors	1	2	3
Volume (L)	91 000	91 000	114 000
Retention time (min)	20	20	25
Iron-to-arsenic ratio		10:1	

Teck-Corona Mine. The Teck-Corona mine is in the Hemlo area. It currently treats about 1000 tonnes of ore per day. The tailings pond area covers about 35 hectares. About 50 to 70 percent of the process water required for the mill is reclaimed from the tailings pond. The tailings pond supernatant is treated to remove cyanide, arsenic,

TABLE 22 PERFORMANCE DATA (CON MINE)

	Constituent (mg/L)			
	Cu	As	CN_{WAD}	CN_T
Influent	3	27	2	3
Effluent	0.15	0.33	0.22	0.28
% Removal	95.0	98.8	89.0	90.7

* For September 1986

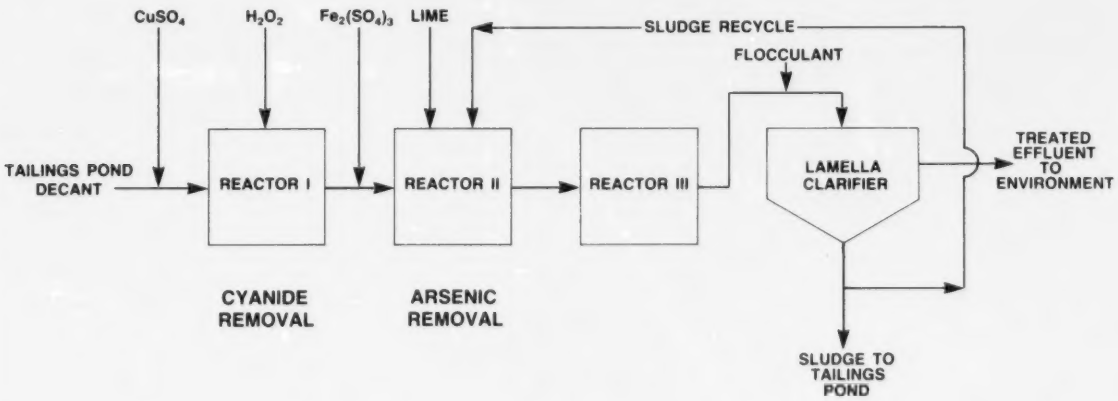


FIGURE 10 CON MINE - EFFLUENT TREATMENT PROCESS

antimony, and heavy metals, including molybdenum, before discharge to Lim Lake (see Figure 11). As the summer season progresses, more and more of the cyanide is removed by natural degradation in the tailings pond and eventually drops below the Ontario guideline concentration. Once this occurs, the cyanide destruction step is discontinued. However, treatment for removal of the other components continues.

The treatment circuit consists of three tanks, each 4 m in diameter and 4.28 m high, and an Eimco clarifier 9.75 m in diameter and 3.55 m high. The first two tanks are for cyanide removal and the third is for the precipitation-coprecipitation reaction to remove arsenic and heavy metals. Hydrogen peroxide and copper sulphate are added to the first tank and lime for pH control to the first two tanks. Iron is added in the form of ferric sulphate to the third tank. Sulphuric acid is added to the clarifier overflow to drop the pH from 8.5 to 7.5 to reduce the toxic effect of ammonia in the effluent. (See Table 23 for operating parameters and Table 24 for performance data.)

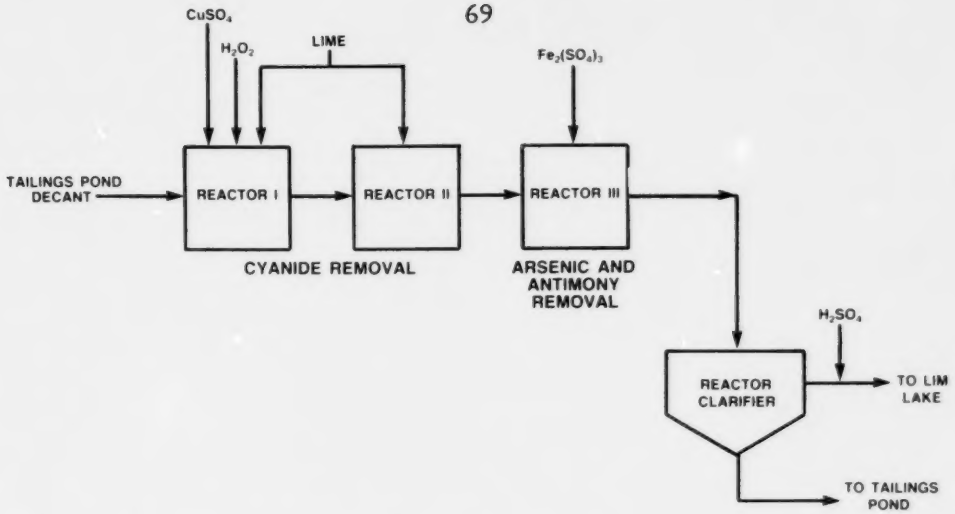


FIGURE 11 TECK-CORONA - EFFLUENT TREATMENT PROCESS

TABLE 23 OPERATING PARAMETERS (TECK-CORONA MINE)

Mill feed rate	1000 t/d
Solution feed rate to treatment plant	100-150 m^3/h
Retention time, per reactor	20 min

TABLE 24 PERFORMANCE DATA (TECK-CORONA MINE)

Note: Removal of cyanide, related compounds and metals based on the averages of weekly grab samples, May 1986.

	Constituent, (mg/L)			
	CN_{WAD}	CN_T	CNO	CNS
Influent	2.35	2.9	36.9	6.4
Effluent	0.10	0.53	42.3	3.5
% Removal	95.7	81.8	---	45.0

	Constituent, (mg/L)					
	Cu	Zn	Fe	As	Mo	Sb
Influent	0.89	0.042	1.10	0.035	0.44	0.047
Effluent	0.47	0.006	1.00	0.005	0.38	0.040
% Removal	47.3	84.5	9.0	86.6	13.2	15.0

5.4.4 Treatment for Radium Removal. The flow diagram for the removal of radium-226 from wastewaters at the Key Lake uranium mine in northern Saskatchewan is shown in Figure 12 (Neven et al., 1985).

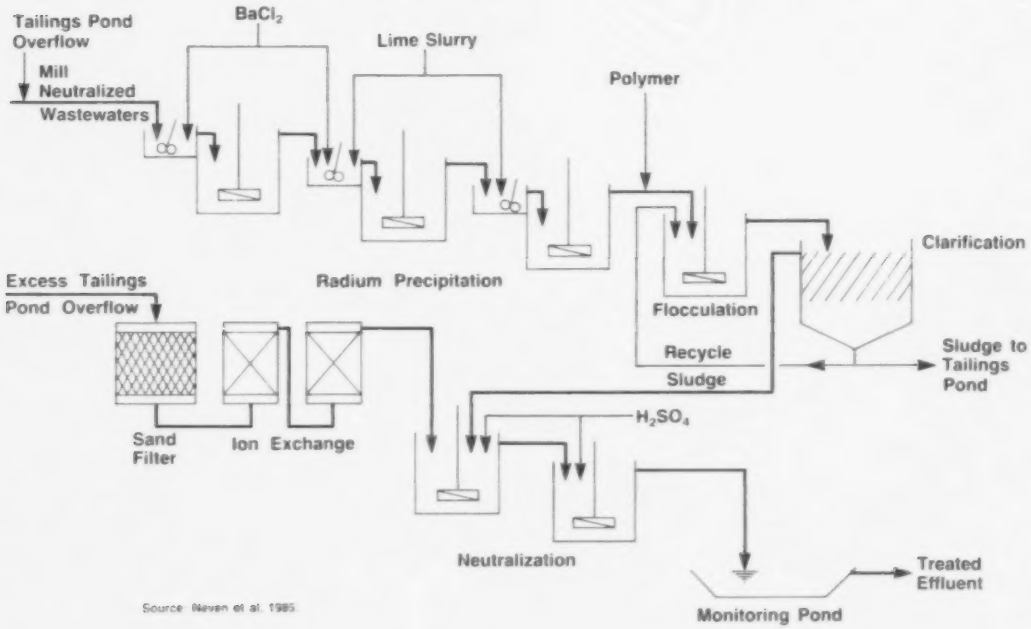


FIGURE 12 KEY LAKE - EFFLUENT TREATMENT PROCESS

Two processes are used for the removal of the residual radium content following bulk neutralization of the mill effluents:

- a conventional precipitation system for the removal of radium from the neutralized mill waste effluents; and
- a standby ion-exchange system for the treatment of excess tailings pond water.

In the regular effluent treatment circuit, radium is co-precipitated with barium sulphate in a series of precipitation tanks. The precipitates are then flocculated before solids-liquid separation using a plate-type clarifier. The clarifier overflow is neutralized by the addition of sulphuric acid and then discharged into a monitoring pond before release from the mining site. The clarifier sludge is disposed of in the tailings pond.

The standby ion-exchange plant provides additional treatment capacity to ensure that low levels of radium activity are achieved. Clear effluent from the tailings

ponds is filtered and then passed through two radium-specific ion exchanges, producing an effluent with less than 0.2 Bq/L of radium-226. This effluent is also discharged into the monitoring pond after neutralization. The exhausted resin is disposed of in the tailings pond. (See Table 25).

TABLE 25 PERFORMANCE DATA (KEY LAKE URANIUM MINE)

	Removal		
	Ra-226 (Bq/L)	As (mg/L)	Ni (mg/L)
Regular - Influent	1 - 5	2 - 3	10 - 20
- Effluent	0.01 - 0.15	0.2 - 0.3	<0.25
Standby - Influent	1 - 4	0.05 - 0.40	0.1 - 0.4
- Effluent	<0.2		

5.5 Treated Water Effluent Quality

A summary of effluent water qualities reported for a number of treatment systems in operation at various Canadian mines are given in Table 26.

TABLE 26 EXAMPLES OF TREATED EFFLUENT QUALITIES AT CANADIAN MINES

	Base Metal Minewater Treatment System	Base Metal Mining-Tailings Pond Overflow	Gold Milling		Uranium Mining Treated Effluent
			Effluent Mine A	Effluent Mine B	
pH	9.5	9.5			
Cyanide (mg/L)			0.3	0.02	
Arsenic (mg/L)		<0.01		0.38	0.2-0.3
Copper (mg/L)	<0.02	<0.05		0.04	<0.05
Iron (mg/L)	1.4	<1.0	0.6		
Lead (mg/L)	<0.05	<0.1		0.02	<0.05
Nickel (mg/L)			<0.1	0.05	<0.25
Zinc (mg/L)	0.2	<0.2	<0.1	0.05	<0.05
Radium-226 (Bq/L)					0.01-0.15

6 WASTEWATER MANAGEMENT AT THE POST-OPERATIONAL PHASE AND THE PERMANENT SHUTDOWN OF MINES

6.1 Introduction

The principal features of most mines, once productive operations have ceased, include the following:

- abandoned underground or open-pit workings;
- waste rock disposal area;
- tailings disposal area;
- general mine facilities including roads, plant site; and
- drainage control and treatment systems.

Requirements to ensure the safety and physical stability of mines on abandonment are normally addressed under the regulations of the applicable provincial or territorial mining acts. In many cases these acts also include provision for reclamation of the site in terms of revegetating those areas disturbed by mining operations.

The most important criterion regarding the need for long-term wastewater treatment at the post-operational phase of a site is the acid-generation potential of the waste rock, tailings, or other exposed surfaces.

6.2 Non-Acid-Producing Sites

At non-acid-producing mines reclamation measures should be instituted so that the mines can be left without the need for long-term wastewater treatment facilities. The objectives of reclamation measures at non-acid-producing sites include:

- prevention of surface erosion by wind or water;
- utilization of the site for agriculture, wildlife habitat or other purposes; and
- establishment of a safe and aesthetically acceptable area.

Achieving these objectives normally requires proper grading, covering and stabilization of slopes and other surfaces that might otherwise lead to high sediment loadings in runoff. Revegetation of disturbed surfaces is often an important element in the reclamation plan and seldom poses undue technical difficulties given reasonable working slopes and the availability of acceptable growth media.

6.3 Acid-Producing Sites

The presence of pyrite and other sulphide minerals in unbuffered waste rock, tailings, and other exposed surfaces, such as pit walls, can result in the production of

contaminated drainage which, if not effectively controlled, may forever require treatment.

The long-term cost of maintaining a drainage control system, lime neutralization treatment plant, and sludge disposal facilities can be extremely high, particularly at remote sites where the maintenance of access for operating staff and reagent delivery can be a major long-term commitment.

The implications of providing drainage control and treatment facilities during the post-operational phase must be considered at the earliest possible stage, so that every practical effort can be made to reduce the volume and strength of drainage requiring treatment. For new mines, these considerations should be made as a function of the mine feasibility and pre-development planning activities.

The objective in planning post-operational waste management measures should be to reduce the volume and strength of acid drainage to a level where passive techniques are sufficient treatment of the residual drainage before discharge. That would eliminate the need for mechanized treatment and its associated problems of maintaining access, reagent delivery, sludge disposal, and maintenance. This objective has yet to be achieved at an acid-producing mine in Canada but the performance of passive treatment techniques, such as algal adsorption, will be evaluated as a function of the Reactive Acid Tailings (RATS) Program (see Section 6.5.2).

To limit the production of acid drainage by adding neutralization reagents such as lime or limestone to waste rock piles or tailings deposits is seldom practical.

6.3.1 Present State-of-the-Art. The current state of the art for managing acid-generating mines at the post-operational phase aims to provide long-term collection and treatment facilities for the acid drainage. Currently no generally applicable methods allow such sites to be abandoned without continuing treatment, if acceptable effluent standards are to be maintained.

The following methods minimize the cost of providing long-term collection and treatment:

- ensure that all potentially acid-generating material is stored under controlled conditions and that its use for the construction of site facilities is rigorously avoided during the construction and operational phases of the mine;
- flood pits and mine workings to the maximum extent feasible to reduce acid production;
- locate waste rock piles in areas where surface drainage and seepage can be controlled;

- where suitable material is available, provide progressive sealing layers in waste rock piles to limit the availability of air and moisture (particularly air) for the oxidation of sulphide material;
- provide a site drainage control system designed and operated to segregate contaminated and clean drainage to the maximum extent feasible. The capacity of the system should be sized on the basis of accepted statistical runoff intensity and duration criteria;
- reduce oxidation of sulphide-containing tailings by covering or flooding disposal area (see Section 6.5.2); and
- treat residual drainage (see Section 6.3.2) before discharge.

6.3.2 Post-Operational Treatment. The normal requirement of post-operational wastewater treatment plants at acid-generating mines is the neutralization of acid and removal of metals contained in the drainage (see Section 4.1). The treatment of process wastes is obviously not an issue at this stage. However, the absence of an active tailings impoundment eliminates, in most cases, the option of disposing of AMD treatment sludges in the tailings pond which is normal practice during the operational phase.

The objectives for managing AMD sludge during post-operation are the same as those during operation:

- minimize the volume of drainage to be treated by segregating contaminated drainage and reducing AMD production as much as possible;
- minimize the volume of sludge through use of the high-density sludge treatment process (see Section 4.1.6); and
- dispose of thickened sludge in a chemically stable environment to prevent dissolution of metals and other constituents.

Options for the ultimate disposal of treatment sludge at the post-operational phase are highly site-specific but they may include the following:

- offsite disposal in landfill or other waste disposal facility;
- on-site landfill;
- on-site lagoons; and
- underwater disposal in flooded pits or mine workings, if alkaline conditions can be assured.

At most acid-producing sites, the volume and strength of contaminated drainage is greater during the operational phase before the application of final reclamation measures, such as the sealing and revegetation of waste dumps and the flooding of pits or shafts. Therefore, estimating the characteristics of the drainage to be treated at the post-operational stage is often difficult.

The recommended approach to overcoming this difficulty is first to apply progressive reclamation techniques to as many sources of contamination as possible throughout the working life of the mine, and second to use a system of site-monitoring which defines the loadings of contaminants from each source (i.e., measures both the volume and concentration of contaminants rather than only concentrations). This not only enables priorities to be established in applying reclamation measures but also provides a means for estimating the loadings of metals and total acidity likely to require treatment once the final reclamation measures have been put in place.

6.4 Uranium Mines

In the case of non-sulphide bearing waste dumps or tailings areas containing radionuclides associated with uranium mining, adequate cover must be provided to achieve a stable surface that will not be eroded by wind or surface runoff.

At the Beaverlodge site (Eldorado City) radionuclide-containing wastes were either removed for disposal in old mine workings or covered with 0.6 m of locally available inert material. Tailings areas not already under water were covered with 0.6 m of rock or removed for disposal under water. Objectives for maximum radiation dose levels were 5 mSv/a for casual visitors to the site and 50 mSv/a for occupational exposure. No treatment facilities for the removal of radium or other radionuclides from wastewater were provided. Similarly, at the Bancroft Mine, a non-acid-producing uranium mine near Peterborough, Ontario, no wastewater treatment facilities for the removal of radium may be required after closure.

At uranium properties that are acid-generating it is necessary to provide treatment for the neutralization of the acid and removal of metals at the post-operational phase as described in Section 6.3. The need to maintain barium chloride addition for the removal of radium is site-specific.

6.5 Abandonment of Tailings Impoundments

Tailings impoundments can be categorized into three, sometimes overlapping groups relative to achieving long-term chemical and physical stability of the deposits at the post-operational stage:

- chemically stable tailings;
- acid-generating sulphide tailings; and
- radioactive tailings.

The principal reclamation issues associated with each category are briefly reviewed in the following sections.

6.5.1 Reclamation of Chemically Stable Tailings. At many mining sites in Canada the tailings are chemically stable and therefore do not produce contaminated seepages or runoff. Usually the surface of the tailings requires only minor additions of fertilizer and other soil ameliorants to enable a self-sustaining vegetative cover to be established. The key issues with such tailings impoundments during the post-operational phase are therefore:

- geotechnical integrity of the impoundment;
- routing of surface water drainage; and
- stabilization of the surface (usually by vegetation) to prevent wind and water erosion.

These issues can usually be addressed adequately at the design stage with the most important consideration being the final physical configuration of the surface of the tailings deposit (see Section 5.1.5) relative to post-operational drainage patterns. The solution to this depends on net precipitation characteristics for the area, decant arrangements in place, and the practicality of creating a stable, free-draining surface.

Where the tailings pond is adjacent to the dam, the installation of a permanent spillway in the dam to provide for surface drainage from the area controlled by the tailings impoundment may be feasible. In other cases it may be necessary to consider erecting a domed-shaped tailings deposit by spigotting tailings from the centre of the impoundment during the final years of operation or by backfilling the area with waste rock or other available material.

6.5.2 Acid-Generating Tailings. With acid-generating tailings, ways to deal with the long-term effects of contaminated seepage and runoff resulting from the oxidation of sulphides in the tailings have to be found.

The approach most commonly used to date has been to establish a vegetative cover with a view to limiting the availability of air and moisture to the tailings. This approach has not been proven as a solution to the acid generation problem although it does fulfill the objective of stabilizing the surface. The approach is often expensive because a zoned covering of permeable material is normally required to prevent the acid in the tailings from rising through the soil cover or surface layer by capillary action and thus destroying the root system of the vegetation.

The present state-of-the-art does not provide any universal solutions for the abandonment of acid-generating tailings. For this reason a 5-year program of applied research, jointly funded by industry and both the federal and some provincial governments, called the Reactive Acid Tailings (RATS) Program, was initiated in 1985 by CANMET. The program has been expanded to deal with problems created by acid-generating waste rocks.

Flooding of acid-generating tailings areas, where feasible, appears to offer a means of curtailing the oxidation reaction and will receive further examination. Otherwise, an extensive surface seal has to be applied to limit access of air and moisture to the tailings and any residual contaminated seepage flows collected and treated.

The design and operation of acid-generating tailings impoundments should recognize the likelihood of having to undertake corrective measures and seek to reduce their long-term cost at the planning stage. Measures to be considered include:

- use of a low-permeability lining to limit seepage;
- use of low-permeability retaining structures made of chemically stable material;
- operation with the pond adjacent to the dam to facilitate flooding;
- minimization of surface area of the impoundment to reduce reclamation costs;
- availability of covering materials if flooding is neither practical nor desirable; and
- methods of passive treatment for residual contaminated seepage.

Ongoing results of the Reactive Acid Tailings (RATS) Program should be reviewed with regard to progress in defining better the options to long-term management of acid-generating tailings.

6.5.3 Radioactive Tailings. Uranium tailings may contain unbuffered sulphides, and thus be acid-generating, as well as radium and other radionuclides. The measures described in Sections 6.5.1 or 6.5.2 must therefore be considered in addition to those directly associated with the long-term control of radiation exposure levels and radionuclide-containing drainage. Documentation published by the Department of Energy, Mines and Resources as a function of the National Uranium Tailings Program, should be consulted.

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